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COLD LAKE - BEAVER RIVER WATER MANAGEMENT STUDY

VOLUME 8
WATER QUALITY

Appendix M .... Lake Trophic Assessment
Appendix N ... Impacts of Lake Level Fluctuations on Trophic Status
Appendix O .... River Water Quality Assessment

Water Resources

Alberta

ENVIRONMENT
Planning Division

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COLD LAKE — BEAVER RIVER
WATER MANAGEMENT STUDY

VOLUME 8
WATER QUALITY

PREPARED BY
PLANNING DIVISION
ALBERTA ENVIRONMENT
December 1983

Submitted to: Hon. F. Bradley
Minister of Environment

APPENDIX M

LAKE TROPHIC ASSESSMENT

A Report Prepared By
D.O. TREW, E.I. YONGE, R.P. KAMINISKI
ALBERTA ENVIRONMENT
POLLUTION CONTROL DIVISION
WATER QUALITY CONTROL BRANCH

Alberta
ENVIRONMENT
Environmental Protection Services
Pollution Control Division

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Sept. 8, 1983

Mr. Ron Hicks
Athabasca River Basin Planner
Planning Division
Alberta Environment
9820 - 106 Street
Edmonton, Alberta
T5K 2J6

Dear Mr. Hicks:

We are pleased to submit the following report entitled "A Trophic Assessment of Nine Lakes in the Cold Lake Region - Interim Report".

The report outlines the baseline chemical and biological data collected on seven lakes in the Beaver River watershed during the 1980 field season. The report also describes the existing phosphorus budgets, both calculated and measured, for these same lakes prior to significant land use changes resulting from industrial development.

Yours truly,

A/DIRECTOR

R. N. Briggs, M. Sc., P. Eng.
for Director
Pollution Control Division

DT/sl

SUMMARY

A baseline assessment of nine lakes in the Cold Lake Region was initiated during 1980. The principal objectives of this investigation are:

(a) to define the existing trophic states of the major recreational lakes of this region, and

(b) to generate estimates of their individual phosphorus loadings so that predictive statements regarding the potential impacts of watershed disturbances can be made, and

(c) to establish baseline chemical data.

All of the lakes studied are naturally productive, and with three exceptions could be classified as meso-eutrophic on the basis of phytoplankton chlorophyll 'a' concentrations. Tucker Lake and the north basin of Primrose Lake are shallow, holomictic waterbodies, and as such display higher fertilities, while Cold Lake appears meso-oligotrophic. A tentative ranking of lakes based upon mean chlorophyll 'a' concentrations suggests the following trophic pattern: Primrose North > Tucker > Ethel > May > Hilda > Primrose South > Moore > Bourque > Marie > Cold. The mesotrophic lakes display autumn phytoplankton blooms, whereas the more productive lakes bloom during summer.

Hypolimnetic oxygen depletion occurs during the summer in all stratified basins, although the depletion rates are variable. Oxygen concentrations decline over winter in all lakes, but Tucker Lake displays a potential winterkill situation; the data suggest that this lake may be particularly sensitive to altered nutrient loadings.

An incubator technique was devised to provide for improved lake to lake comparability of productivity measurements. The productivity data, as determined by the radiocarbon uptake method, are highly correlated with standing crop observations and the two have been combined to determine a mean photosynthetic index for each lake. This index has been useful in demonstrating the potential interaction of both phosphorus and nitrogen in phytoplankton carbon fixation in these lakes.

Algal bioassay data on all lakes suggest that a small fraction of the total phosphorus pool is bioavailable (approximately 25-35%), and most of the latter is found in the particulate form. In the two tributary streams assessed a higher portion of biologically-available phosphorus was found (45 - 55%); most of this was in the dissolved fraction. In-lake nitrogen is predominantly present as the organic fraction; nitrate and nitrite are usually close to the detection limit and ammonia is variable but generally low.

Further refinements to the phosphorus modelling sequence as a management tool on productive lakes have been initiated. Whole-lake chlorophyll surveys were conducted on four lakes to quantify phytoplankton spatial heterogeneity at the community level. The results suggest that the sampling strategy applied during 1980 may have been sufficient to define representative whole-lake nutrient and trophic characteristics. The problem of defining accurate total phosphorus loadings when internal vectors are present has been partially addressed by monitoring hypolimnetic phosphorus buildup in Ethel Lake. The data suggest that at fall overturn there is a net decrease in water column phosphorus content to pre-stratification levels. This work is being extended during 1981 to other stratified basins.

ACKNOWLEDGEMENTS

We acknowledge the assistance of several branch staff members in various stages of this work. We thank R. Nelson for extensive field and lab assistance; D. Bespalko for processing the production and standing crop samples; H. Hamilton and B. Brown for providing the algal growth potential and BAP data; T. B. Reynoldson for statistical advice; G. Bodie, J. Kostler, C. Campbell, C. S. Liu and all of the above for occasional field assistance. A. Makin typed the manuscript.

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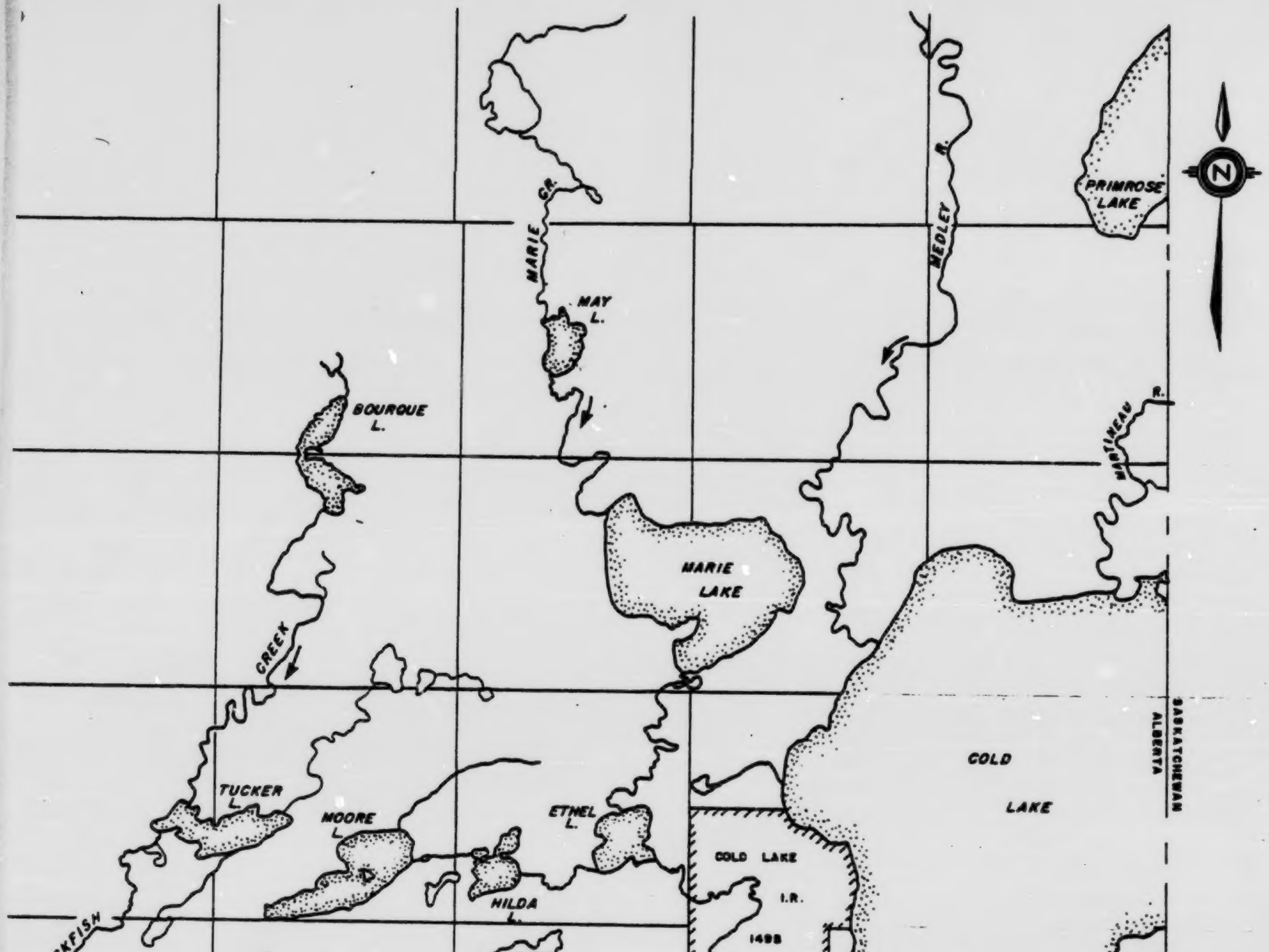
INTRODUCTION

A synoptic survey of seven lakes in the Jackfish and Marie Creek drainages was initiated during 1979/80; a preliminary analysis of the 1980 data is presented in this report. Most of the methods used are summarized in detail elsewhere (Trew et al., 1981) and will not be discussed here. All analyses were performed on integrated epilimnetic samples combined from 20 sites in each lake. Symbols generally follow the recommendations of Winberg (1971).

7

The seven lakes investigated were May, Marie, Ethel, Hilda, Moore, Bourque, and Tucker (Figure 1). In addition, an arrangement was made with Environment Canada, Water Pollution Control Branch to monitor Cold and Primrose lakes jointly; some comparative data from the latter are also included in this report. Sampling was conducted at fortnightly intervals during the open-water season, although Cold and Primrose Lakes were sampled over a shorter time period (June - September).

The lakes lie in a rolling upland area of the boreal mixed-wood forest containing a substantial amount of muskeg. The watersheds of most lakes are essentially undeveloped. The dominant tree species are those found throughout the lakeland area: white spruce, trembling aspen, and jack pine in the well drained soils, with black spruce and tamarack characteristic of the low lying sites; some birch and balsam poplar are also evident. The area is underlain by Upper Cretaceous bedrock and is covered by glacial till ranging up to two hundred meters in thickness.


PRIMROSE LAKE
N
MARIE CR.
MAY L.
MEDLEY R.
MARTINEAU R.
BOURQUE L.
MARIE LAKE
CREEK
TUCKER L.
MOORE L.
ETHEL L.
HILDA L.
JACKFISH
COLD LAKE I.R. 149B
COLD LAKE
SASKATCHEWAN
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COLD LAKE INDIAN RESERVE

FIGURE I STUDY AREA

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THE PHYSICAL ENVIRONMENT

MORPHOMETRY AND HYDROLOGY

The principal morphometric and hydrologic features of the study lakes are summarized in Table 1. These data have been supplied by the Hydrology Branch, Technical Services Division, Alberta Environment (R. Bothe, personal communication).

The surface drainage areas presented in this table exclude upstream basins. Many of the hydrologic parameters will be modified over the next few years as new data are incorporated; an intensive hydrologic network was planned and partially installed during 1980 as a result of requirements of the Cold Lake - Beaver River Regional Water Resources Study (Alberta Environment, 1980).

The most obvious feature about these lakes is that they receive low areal water loads - runoff is low in this area and the effective watershed areas are also relatively small. However, due to their small mean depths, lakes such as May and Ethel still display short residence times. In all cases, precipitation inputs represent a significant portion of the known total annual inflow. The role of groundwater in the water budgets of these lakes is not understood at present, however a modelling study using the technique developed by Crowe (1979) on Baptiste Lake is being undertaken by Earth Sciences Division, Alberta Environment with the view to addressing this problem.

TABLE 1. MORPHOMETRIC AND HYDROLOGIC CHARACTERISTICS

	May	Marie	Ethel	Hilda	Moore	Bourque	Tucker
Volume (dam^3)	16870	510540	32021	22604	77260	41921	21133
Lake Surface Area (km^2)	3.02	36.21	4.83	3.62	9.30	5.00	7.22
Surface Drainage Area (km^2)	64.09	171.22	34.02	37.2	32.84	112.57	149.53
Mean Depth (m)	5.6	14.1	6.6	6.2	8.3	8.3	2.9
Annual Precipitation (dam^3)	1298	15570	2077	1557	4007	1997	2557
Total Annual Inflow (dam^3)	12601	35202	20153	3838	5680	7558	15992
Annual Outflow (dam^3)	10880	14562	17400	1755	380	4377	11935
Residence Time (years)	1.34	14.5	1.59	5.89	13.6	5.5	1.3
Flushing Time ($year^{-1}$)	0.65	0.03	0.54	0.08	.01	0.10	0.57
Areal Water Load (m)	4.17	0.97	4.17	1.06	0.61	1.51	2.22

MIXING REGIMES

Temperature, oxygen, conductivity and light profiles were recorded at the deepest point in each lake. In many cases these locations were not representative of the entire water body, with many lakes having both stratified and unstratified areas; however, time constraints allowed for only one station per lake. Certain physical characteristics of the water column area probably similar in areas of similar depth, allowing for limited extrapolation when necessary.

These lakes would generally be described as dimictic, although partial meromixis is becoming increasingly more apparent in Alberta as more lakes are examined. This incomplete mixing usually occurs in the spring as a result of rapidly increasing air temperatures in May coupled with low winds. Furthermore, the morphometry of many of these study lakes is such that most of the deeper basins have little fetch and lie in close proximity to landforms which provide shelter from prevailing winds. Incomplete mixing was in fact observed in several of these lakes during the spring of 1980.

The mixing regimes for all lakes have been summarized in Table 2. Tucker Lake and the North basin of Primrose Lake were holomictic during the open water season, but all other lakes appeared to stratify. Ethel, Bourque, Moore, and May were only partially mixed during spring, with the northern basin of Hilda perhaps not having mixed at all (see below). Marie lake was completely and quickly mixed due to its large size and exposure. Spring and

TABLE 2. SUMMARY OF MIXING REGIMES

LAKE	TYPE	SPRING MEROMIXIS	MAX. EPI TEMP. (°C)	THERMOCLINE DEPTH (m)	ISOTHERMAL DATE
Tucker	Holomictic	No	20.0 (July 8)	---	---
Ethel	Dimictic	Yes	19.9 (July 23)	9-12	October 29
Bourque	Dimictic	Yes	19.4 (August 18)	9-11	" 28
Moore	Dimictic	Yes	19.5 (July 21)	9-13	" 14
May	Dimictic	Yes	19.0 (July 23)	7-10	" 14
Hilda	Dimictic	Yes	20.9 (July 22)	6-8	" 15
Marie	Dimictic	No	18.9 (July 22)	9-12	" 16
Primrose North	Holomictic	?	20.0 (July 12)	---	---
Primrose South	Dimictic (?)	?	17.0 (August 10)	12-16	?
Cold	Dimictic (?)	?	18.0 (July 14)	Variable	?

autumn observations are not available on Primrose and Cold Lakes, consequently their mixing characteristics cannot be completely defined.

The maximum summer epilimnetic temperatures ranged from 17.0°C to 20.9°C and occurred in early July on Tucker, Cold, and Primrose North, and at the end of July in all other lakes except Bourque and Primrose South in which it was delayed until mid-August. Isothermal conditions were first observed in Marie Lake by mid-September; most other lakes became isothermal by mid-October, however, Bourque and Ethel remained stable until the end of October.

The vernal mixing provided a thorough oxygen replenishment in both Tucker and Marie lakes. However, the incomplete turnover in Ethel, Bourque, May and Moore is readily apparent in Figure 2 with oxygen concentrations in early May being well below saturation; no hypolimnetic oxygen was ever recorded in Hilda Lake. The hypolimnia of all lakes (except Marie and Primrose South) were devoid of oxygen by the end of June.

All lakes experienced overwinter oxygen deficits. however the sampling frequency prohibited the calculation of rates. Tucker Lake was the most severely affected with late winter (March) surface concentrations falling to a range of 1-3 gm O_2 m^{-3}.

The summer hypolimnetic oxygen depletion rates have been calculated according to the technique of Lasenby (1975) and are expressed both on an areal basis (AHOD) and a volumetric basis (VHOD). The top of the hypolimnion was defined as the second inflection point on the temperature profile in early August, and a constant hypolimnetic volume was assumed; volume-weighting was

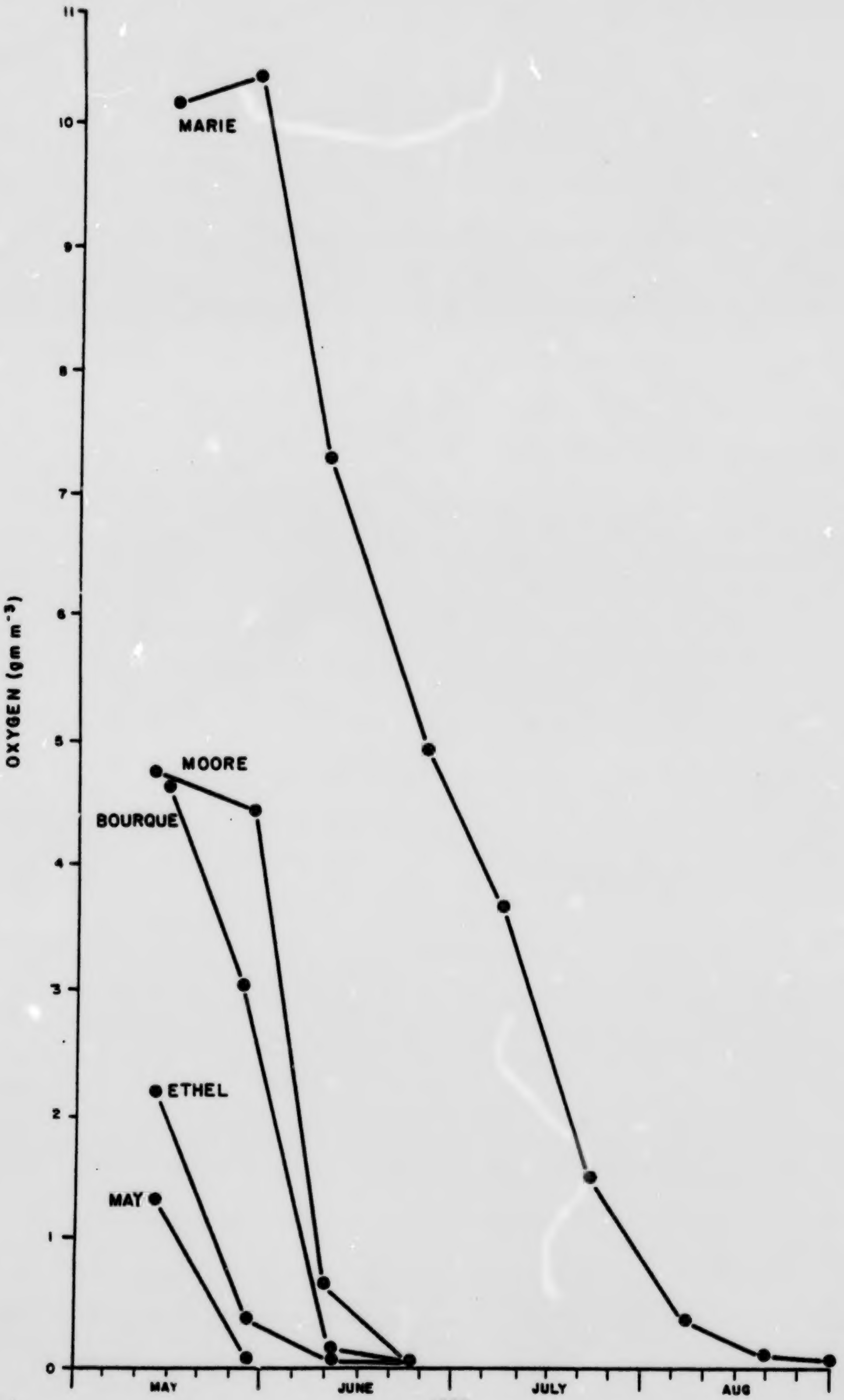

FIGURE 2 VOLUME-WEIGHTED HYPOLIMNETIC OXYGEN CONCENTRATION

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applied to the data. The rates were determined on the most linear portion of the depletion curve (Barica and Mathias, 1979), and are summarized in Table 3 along with the mean hypolimnetic depth (z_H).

TABLE 3. SUMMARY OF SUMMER OXYGEN DEPLETION DATA

LAKE	DEPLETION DATE	AHOD (gm m^{-2} day^{-1})	VHOD (gm m^{-3} day^{-1})	z_H (m)
Ethel	June 24	.964	.130	7.4
Bourque	June 24	1.586	.165	9.6
Moore	June 23	1.666	.271	6.2
May	May 27	.265	.095	2.8
Hilda	May 12	---	---	2.9
Marie	August 20	.923	.145	6.4

Cornett and Rigler (1980) have recently reviewed the literature dealing with areal hypolimnetic oxygen deficits and have concluded that the AHOD model does not remove the influence of lake morphometry on inter-lake comparisons as originally proposed by Hutchinson (1938). The data on these lakes are preliminary and will not be manipulated from a predictive viewpoint until the data set is expanded. These depletion rates are, however, at the upper end of the range presented by Cornett and Rigler (1980).

LIGHT PENETRATION

Detailed light observations were made on all lakes. Extinctions were calculated as:

$$\varepsilon = \frac{\ln I_o - \ln I}{z}$$

where ε = vertical extinction coefficient (ln units m^{-1})
I_o = initial light intensity
I = intensity after passing through a water column z meters thick.

The mean euphotic zone extinction data for all lakes are summarized in Table 4; also included are the mean euphotic zone depths (z_{eu}) as determined by the depth of 1% surface penetrating irradiance. The latter are estimated to the nearest half meter.

TABLE 4. SUMMARY OF MEAN LIGHT DATA

LAKE	$\underline{\varepsilon}$	$\underline{z}_{eu}$	$\underline{z}_{eu} \cdot \varepsilon$
COLD	.272	15.5	4.2
PRIMROSE (SOUTH)	.400	9.5	3.8
MOORE	.507	9.0	4.5
MARIE	.510	9.0	4.6
BOURQUE	.565	8.0	4.5
ETHEL	.641	7.5	4.8
HILDA	.660	6.5	4.3
MAY	.825	5.5	4.5
TUCKER	1.175	4.0	4.7
PRIMROSE (NORTH)	1.513	3.0	4.5
			$\bar{x} = 4.4$

The relationship between ε and z_{eu} is illustrated in Figure 3. Theoretically, this relationship should be linear since ε is expressed as a natural logarithm. However the data from Cold Lake (lowest extinction) and Primrose Lake (highest extinction) are based solely upon the summer period, and may not be representative of ice-free seasonal means.

Talling (1965) has provided an empirical relationship for defining the depth of the euphotic zone in a wide variety of lake types:

$$z_{eu} = \frac{3.7}{\varepsilon_{min}}$$

Based upon our preliminary observations, and noting that we have measured ε over a band width >>PAR (photosynthetically available radiation taken as 400-700 nm) as opposed to ε_{min}, the product $z_{eu} \cdot \varepsilon$ may be equivalent to 4.4. Therefore, the depth of the euthotic zone may be estimated from the following relationship:

$$z_{eu} = \frac{4.4}{\varepsilon}$$

In addition, Talling (1965) expressed light extinction as an additive function with respect to phytoplankton standing crop and all other non-algal sources:

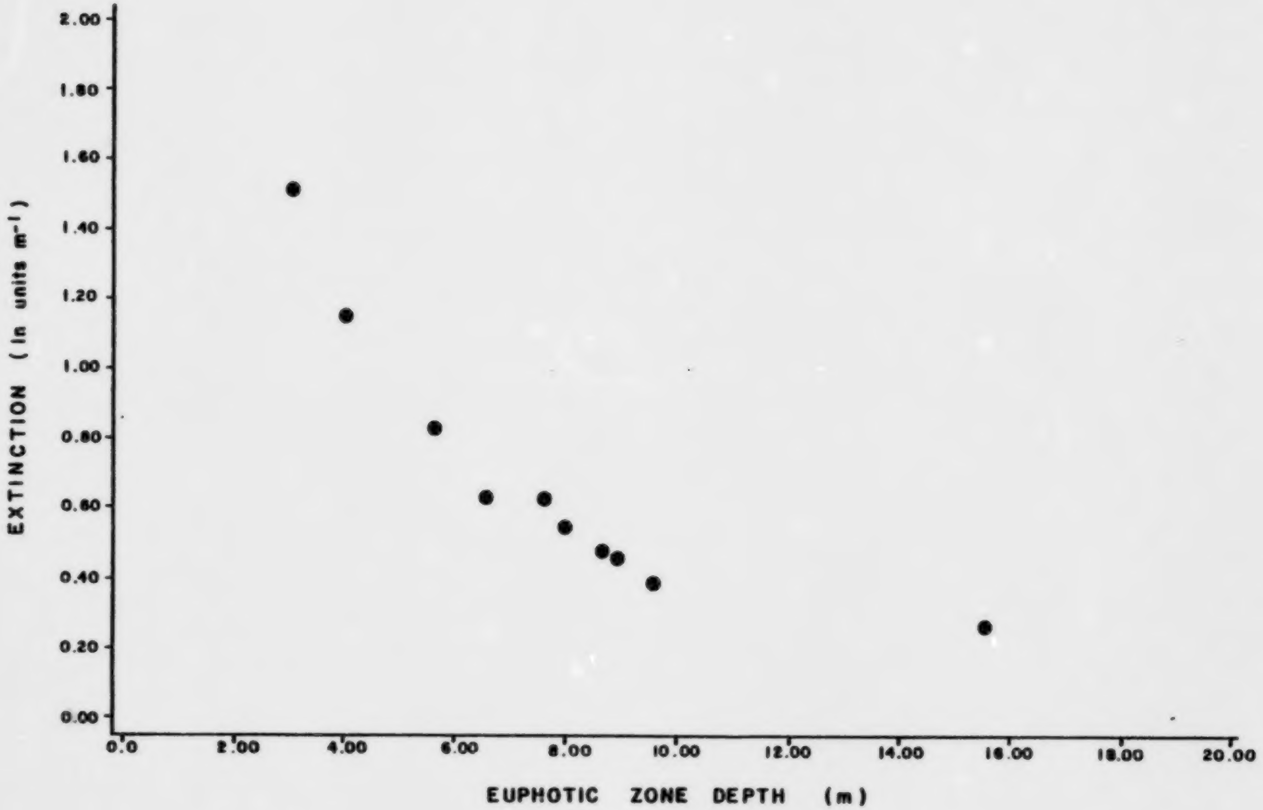

FIGURE 3 MEAN LIGHT EXTINCTION VERSUS MEAN EUPHOTIC ZONE DEPTH

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$$\varepsilon_{min} = \varepsilon_s B + \varepsilon_q$$

where ε_{min} = minimum vertical extinction coefficient (ln units m^{-1})

ε_s = incremental increase in extinction per unit chlorophyll (ln units $mg^{-1}\ m^2$)

B = chlorophyll concentration of the euphotic zone (mg m^{-3})

ε_q = background extinction due to non-algal sources (in units m^{-1}).

By measuring ε over the same band width (>>PAR), a relationship was derived to express light extinction as a function of chlorophyll in Ethel Lake (Fig. 4):

$$[\varepsilon] = 0.027[B] + 0.382$$

$$r^2 = 0.57$$

The slope of this line, 0.027, is identical to that observed in Baptiste Lake (Trew et al., 1981) and similar to others found in the literature: 0.03 - Cooking Lake, Alberta, Hickman, 1979 (a); 0.03 - Ministik Lake, Alberta, Hickman, 1979 (b); 0.02 - Lake Windermere, England - Talling, 1960; 0.02 Lake George, Uganda - Ganf, 1972; and 0.016 - Lake Vombsjon, Sweden - Gelin, 1975.

The background extinction (0.382 ln units m^{-1}) is, however, comparatively low indicating the relative clarity of this lake. The ε_q values for the above lakes range from 0.636 (Baptiste) to 2.55 (Lake George).

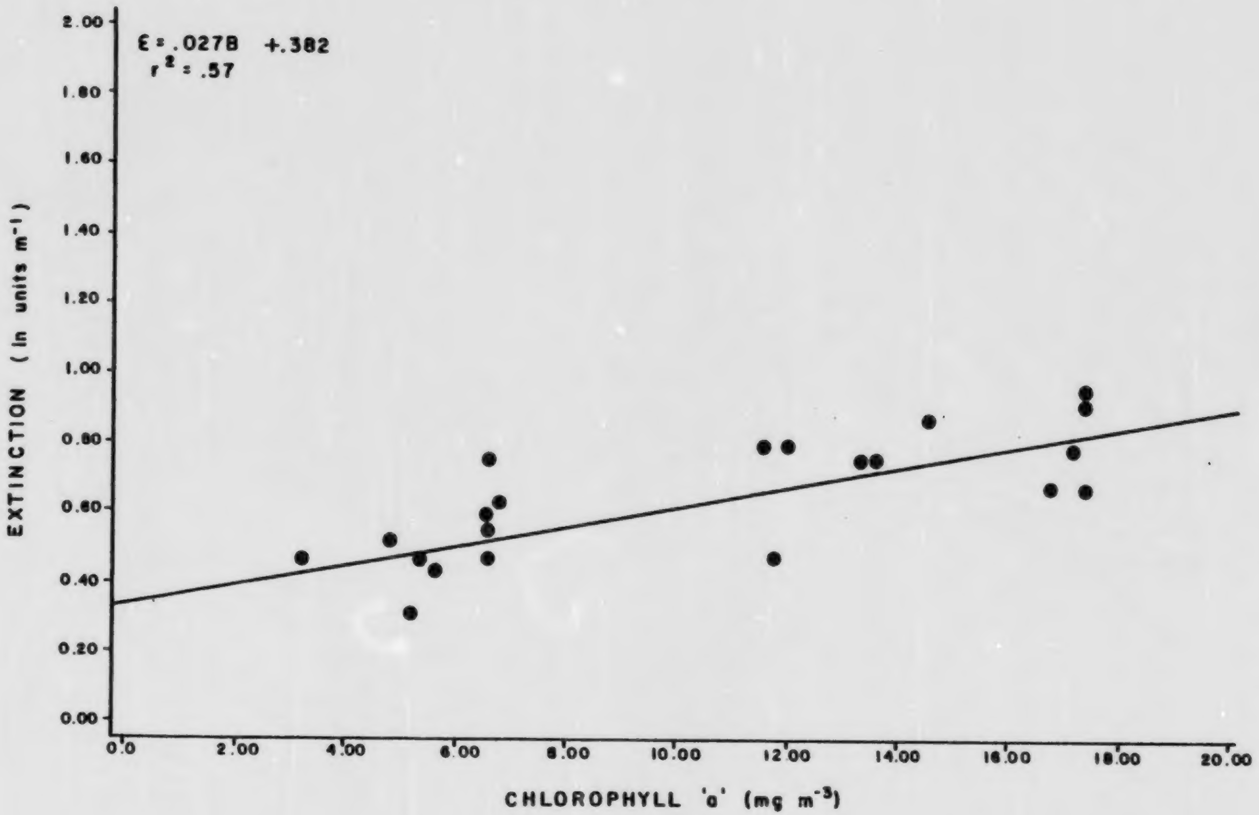

FIGURE 4 LIGHT EXTINCTION AS A FUNCTION OF CHLOROPHYLL 'a' CONCENTRATION IN ETHEL LAKE

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THE CHEMICAL ENVIRONMENT

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Significant differences in ionic composition and total chemical content were evident in these lakes, however the interactions of geological and hydrological features in determining these characteristics have yet to be defined.

The mean specific conductances (at 25°C), mean total dissolved solids, and mean salinities (taken as the sum of Ca^+, Na^+, Mg^+, K^+, HCO_3^-, Cl^-, and SO_4^-) during the open water season are presented in descending order in Table 5. A greater than threefold range in each parameter was observed, with Moore and Hilda lakes displaying substantially higher values than the other lakes. By comparison, the Marie Creek drainage is characterized by low conductivities with Ethel Lake reflecting its downstream reception of waters of both low and high salt content. The waters of the Jackfish Creek drainage are of an intermediate salinity.

These lakes lie close to the Western Canada zone of saline lakes (salinities > 500 mg l^{-1}) as described by Northcote and Larkin (in Frey, 1966), however the majority could be defined as "freshwater lakes of the forest zone" (salinities 200-500 mg l^{-1}). Salinity data are also available on four other lakes in this region: Muriel - 825 mg l^{-1}, Garner - 751 mg l^{-1}, Skeleton - 292 mg l^{-1} (Mitchell, 1979) and Moose - 433 mg l^{-1} (Gallup, 1977).

TABLE 5. MEAN CONDUCTANCE (μmhos cm^{-1}), TDS (mg l^{-1}), & SALINITY (mg l^{-1})

LAKE	CONDUCTANCE	TDS	SALINITY
MOORE	717	397	575
HILDA	674	376	548
TUCKER	387	204	323
BOURQUE	337	184	283
ETHEL	322	173	266
COLD	259	142	218
MARIE	256	140	214
MAY	250	138	208
PRIMROSE NORTH	221	126	190
PRIMROSE SOUTH	222	124	191

The major cations and anions are summarized in Table 6 as open-water mean values. Most lakes are of the calcium-bicarbonate type with the exception of Moore and Hilda, which are dominated by sodium-bicarbonate. Furthermore, the entire pattern of ionic dominance is altered in these two systems; their cation sequence is $Na^+ > Mg^+ > Ca^+ > K^+$, whereas in all other lakes it is $Ca^+ > Mg^+ > Na^+ > K^+$. Similarily, the anion sequence elsewhere is $HCO_3^- > SO_4^- > Cl^-$, however in these two lakes it is $HCO_3^- > CL^- > SO_4^-$. Since the salt concentrations in these lakes are well below saturation, evaporation-precipitation processes are not likely to be the cause of any chemical differences.

Recent work by Ozoray et al., (1980) has provided chemical data on the groundwater reserves of the Sand River Area. In the Moore Lake area, a shift from calcium-magnesium to sodium-potassium cation dominance in the groundwater was observed; this coincides precisely with the composition of the surface waters at the same location and provides a strong suggestion that the hydrological regimes of Moore and Hilda lakes are heavily influenced by groundwater.

TABLE 6. MAJOR ANIONS AND CATIONS (mg l^{-1})

LAKE	HCO_3^-	SO_4^-	Cl^-	Ca^+	Mg^+	Na^+	K^+
MOORE	386	18.2	20.4	14.9	40.5	88.3	6.6
HILDA	371	16.1	21.2	18.9	37.9	76.1	6.8
TUCKER	244	4.4	1.8	27.8	23.1	19.1	3.4
BOURQUE	214	3.3	0.9	33.8	18.4	10.7	2.5
ETHEL	196	3.8	3.5	28.7	16.1	15.6	2.7
COLD	160	4.1	1.1	31.5	10.7	8.7	1.9
MARIE	160	3.1	0.9	30.2	11.8	5.5	1.9
MAY	154	3.3	1.0	32.0	10.4	5.8	1.1
PRIMROSE N.	140	3.6	1.0	22.0	8.4	6.8	1.7
PRIMROSE S.	141	3.5	1.0	28.4	8.4	6.6	1.3

NUTRIENTS

An extensive base of information was assembled on the phosphorus content of these lakes using both chemical and algal bioassay techniques (H. Hamilton and B. Brown, personal communication). Unfortunately, a switchover in laboratories had to be made in mid-July due to the relocation of the government facility. This has introduced an uncertainty to the data and consequently limited the usefulness of seasonal interpretations. The nutrient data will therefore be presented in the form of open-water means which will provide for some comparability since all samples from all lakes will have been treated similarily.

The various phosphorus fractions are summarized in Table 7. Total phosphorus in most lakes was found to be in the range of 18 - 36 $\mu g\ l^{-1}$. However, Tucker Lake and the north basin of Primrose Lake which are both shallow, continuously mixed water bodies were found to have much higher concentrations (79 and 54 $\mu g\ l^{-1}$ respectively). Approximately 20-30% of the total phosphorus pool was found to be available, and most of this was in the particulate form. The portion of the dissolved phosphorus fraction found to be available was usually less than 10%; in addition, the dissolved biologically available phosphorus was always much lower than SRP, further evidence that the latter measurement overestimates the true ortho-phosphate component (Rigler, 1968).

The relationship between total biologically-available phophorus and total phosphorus can be defined as:

$$\mathrm{LOG[TBAP]} = 1.147\ \mathrm{LOG[TP]} - 0.832$$

$$r^2 = 0.88$$

TABLE 7. PHOSPHORUS FRACTIONS (μg l^{-1})

LAKE	TP	TBAP	TDP	DBAP	PP	PBAB	SRP
TUCKER	79	26	36	5	43	21	12
PRIMROSE NORTH	54	10	22	1	39	8	6
HILDA	36	9	21	1	15	8	7
MAY	34	10	20	1	16	8	8
MARIE	30	6	20	1	10	4	10
MOORE	29	8	18	1	14	6	5
ETHEL	27	8	17	1	12	6	6
BOURQUE	25	6	16	1	10	4	8
COLD	20	4	15	2	8	2	3
PRIMROSE SOUTH	18	4	13	1	10	3	3

The total nitrogen pool in these lakes is dominated by the organic fraction, with the inorganic component generally amounting to less than 5% (Table 8). Nitrate was frequently below the detection limit, with ammonia generally being found in the 20-100 μg l^{-1} range. Ammonia appeared to increase in certain lakes in early autumn, possibly the result of hypolimnetic erosion, however these observations were scattered and too variable to define a common trend.

TABLE 8. NITROGEN FRACTIONS (mg l^{-1})

LAKE	TN	TIN	TON	TKN	NH_3	NO_3
TUCKER	1.597	.109	1.486	1.500	.097	.012
PRIMROSE NORTH	1.530	.050	1.475	1.496	.034	.016
HILDA	1.512	.056	1.454	1.463	.049	.007
MOORE	1.195	.039	1.154	1.163	.032	.007
ETHEL	0.996	.037	0.957	0.966	.030	.007
MAY	0.955	.035	0.921	0.930	.025	.010
PRIMROSE SOUTH	0.944	-	0.896	0.970	.024	-
BOURQUE	0.806	.028	0.779	0.780	.026	.002
MARIE	0.720	.046	0.679	0.680	.040	.006
COLD	0.550	.018	0.533	0.540	.015	.003

The shallow lakes displayed the highest total nitrogen concentrations, although Moore and Hilda were also high unlike the pattern for total phosphorus. The large, deep lakes (Marie and Cold) had the lowest concentrations.

An analysis of nitrogen:phosphorus ratios based upon seasonal means was carried out in order to gain insights into the potential for nutrient limitation in these lakes (Table 9). All total N:P ratios suggest phosphorus limitation, based on the simple stoichiometric requirements of the phytoplankton community (Rast and Lee, 1978). However, the ratios of TIN:SRP and TIN:TBAP generally point to nitrogen deficient conditions. This dichotomy suggests that the use of TN:TP ratios on Alberta lakes as an index of relative nutrient availability could be misleading; this problem will remain until some understanding of the availability of the organic nitrogen pool is acquired.

TABLE 9. NITROGEN:PHOSPHORUS RATIOS

LAKE	TN:TP	TIN:SRP	TIN:TBAP
BOURQUE	32.2	3.4	4.5
TUCKER	20.2	9.3	4.3
MOORE	41.2	8.2	5.1
HILDA	42.0	8.3	6.4
ETHEL	36.9	6.5	4.9
MARIE	24.0	4.1	6.8
MAY	28.1	4.3	3.4
PRIMROSE NORTH	28.3	9.2	5.5
PRIMROSE SOUTH	55.2	-	-
COLD	27.5	5.7	4.3

TOTAL PHOSPHORUS LOADINGS

The study lakes are in close proximity to the proposed Esso Resources Cold Lake Heavy Oil Project, consequently surface disturbances and land use changes can be expected to occur in specific watersheds. Furthermore, the permanent regional population is expected to increase dramatically once this and other heavy oil projects are approved. One therefore anticipates impacts on these lakes associated with alterations to either the phosphorus loading or hydrological regimes.

The phosphorus modelling sequence as described by Dillon and Rigler (1975) offers an approach by which these impacts can be assessed. This sequence consists of three basic components:

(a) the prediction of algal standing crop from a knowledge of in-lake phosphorus concentration,

(b) the determination of mean in-lake phosphorus concentration either through observation, or by modelling with a knowledge of the phosphorus loading, hydrology and morphometry of the water body in question, and

(c) the determination of phosphorus loading to the water body either by observation, or by modelling with aid of known nutrient export coefficients and atmospheric deposition coefficients.

A major portion of our current work is directed at refining this technique for application to the naturally productive lakes found in central Alberta.

As a first step, we have undertaken to generate total phosphorus loadings for each water body. However, the determination of an accurate total phosphorus loading is difficult in lakes where groundwater, sediment release, and sediment resuspension vectors exist (Trew et al., 1981). We have therefore decided to generate a range of possible inputs for each lake using three different approaches.

Theoretical loadings (columns D and E, Table 11) have been calculated using the export coefficients presented by Rast and Lee (1978); detailed work carried out on the watershed of Baptiste Lake during 1976-79 has tentatively verified the use of these coefficients for the boreal mixed-wood forest biome in Alberta (Trew et al., 1981). Phosphorus input via atmospheric deposition was calculated using the coefficient of 30.4 mg m^{-2} developed in the same study.

A back-calculation technique utilizing the steady state model of Dillon and Rigler (1974) has also been employed:

i.e. $$[P] = \frac{L\,(1-R_p)}{z \cdot \rho_w}$$

or $$L = \frac{[P] \cdot z \cdot \rho_w}{1-R_p}$$

where $[P]$ = Spring total phosphorus concentration (mg m^{-3})

L = areal load (mg m^{-2} yr^{-1})

Rp = retention coefficient

z = mean depth (m)

ρ_w = flushing rate (yr^{-1})

We have used the mean epilimnetic phosphorus concentration during the open water season instead of spring phosphorus since most of these lakes did not undergo a thorough vernal mixing. As a result, phosphorus would not have been uniformly distributed down the water column making a representative whole-lake estimate difficult.

The estimation of phosphorus retention for use in this model is problematic. Ostrofsky (1980) presents a model which purportedly provides improved estimates over that of Kirchner and Dillon (1975) at low areal water loads (q_s). In our experience, Ostrofsky's model does not accomplish this task at the low q_s values frequently observed on Alberta lakes. In fact, the full range of q_s values for the study lakes (0.061 - 4.17 m yr^{-1}) produces a narrow retention range of 0.745 - 0.768 when used in Ostrofsky's formula; intuitively, one might anticipate much higher retentions at very low areal water loads. By comparison, the model of Kirchner and Dillon (1975) produces a wider range of retention estimates from 0.760 to 0.943 (Table 10).

TABLE 10. PHOSPHORUS RETENTION ESTIMATES

LAKE	Ostrofsky	Kirchner and Dillon
MAY	.766	.760
MARIE	.764	.910
HILDA	.763	.905
MOORE	.768	.943
BOURQUE	.758	.870
TUCKER	.745	.820
ETHEL	.766	.760

Since we lack the criteria to justifiably choose either approach, both retention models have been incorporated into the back-calculated (columns A and B, Table 11) and theoretical loading estimates.

Another theoretical approach discussed by Rast and Lee (1978) and used by them to verify loadings has been used as a third method of determining inputs. This technique compares the mean in-lake phosphorus concentration: influent total phosphorus concentration ratio with an expression of the hydraulic residence time, e.g.:

$$\frac{P}{P_i} = \frac{1}{1 + \tau_w}$$

or

$$P_i = P\,(1 + \tau_w)$$

where P = steady state total phosphorus concentration (mg m^{-3})

P_i = influent total phosphorus concentration (mg m^{-3})

τ_w = hydraulic residence time (yr)

An estimate of the total phosphorus input (column C, Table 11) can then be derived by combining P_i with Q, the total annual inflow.

TABLE 11. ESTIMATED TOTAL PHOSPHORUS LOADINGS (kg yr^{-1})

	A (O)	B (K+D)	C	D (O)	E (K+D)
MAY	1598	1558	920	1113	1000
MARIE	1986	5106	5069	3222	3201
ETHEL	1835	1937	1229	1588	1003
HILDA	272	681	472	713	593
MOORE	93	381	772	683	683
BOURQUE	440	800	634	1264	1264
TUCKER	3732	5220	2703	2119	1895

Upstream retention did not need to be estimated for Moore and Bourque since these are headwater lakes. For most lakes an approximate two-fold range of loadings has been generated; the reasons for this are diverse. The valid application of export coefficients to areas of extensive muskeg such as are found in these watersheds is unproven. Moreover, the final theoretical loading figures (D and E) ignore the potentially significant internal loadings. In addition, the steady state approach is based upon several assumptions regarding loading rates, mixing regimes, and fluctuations in concentrations many of which may not be applicable to productive, dimictic, temperate lakes.

The range of loadings derived with the steady state model is largest (in relative terms) on those lakes with the smallest areal water loads; this is because the retention estimates are widely divergent at low values of q_s. Moreover, the probable interaction of ground and surface waters in Moore and Hilda lakes suggests that the water renewal estimates for those lakes are questionable in themselves.

The theoretical loadings for May, Marie, Ethel, and Tucker lakes were generally below those derived with the steady state model, which may suggest the presence of unquantified, internal sources. The calculated loading for Bourque Lake, however, was substantially higher than the steady state estimate which may indicate an over-estimated loading. This, in fact, is highly possible since the watershed of Bourque Lake has the largest proportion of wetlands and muskeg of any of the study lakes, a feature which would serve to minimize phosphorus export.

Because of the importance of the total phosphorus loading figure in the application of the phosphorus modelling sequence, further work is planned during 1981 to refine these data. Specifically, a major effort will be directed at estimating a representative, whole-lake phosphorus accumulation figure using sedimentation rates derived by ^{210}Pb dating of sediment cores taken at various locations throughout each lake. This figure will be combined with the long-term loss of phosphorus through the outlet in the simple mass balance.

For a given time period:

$$I = \Delta M + S + O$$

where I = total inputs

ΔM = change in mass in the lake

S = losses via sedimentation

O = losses via outlet

ΔM is assumed to be minimal in undisturbed watersheds over the short to medium term.

In addition, the acquisition of data regarding the significance of groundwater in the hydrological budgets of these lakes should enhance the loading estimates derived with the steady state models. The retention estimates, however, will remain questionable until general improvements to the retention models are available.

In order to provide an overall perspective to the range of calculated loadings, an attempt to measure all inputs on one lake was undertaken. Ethel Lake was selected for this study for two principal reasons: ease of access to tributaries, and because of its position as the "sump" for the entire Marie Creek drainage system.

Loadings were measured in streams "M-E" (from Marie Lake) and "H-E" (from Hilda Lake). Inputs via atmospheric deposition were measured using snow and rain collectors; the methods used are described in detail elsewhere (Trew et al., 1981). Loadings were calculated from diffuse runoff areas, urban areas, and domestic/recreational sources, also as described in the previous report. The hypolimnetic build up in total phosphorus was monitored throughout the open water season by sampling at 5m intervals, and volume-weighting the observed concentrations.

The total input from "H-E" (Table 12) during the water year of October 31, 1979 to November 1, 1980 was higher than that from "M-E" although the annual discharges were almost identical ("H-E" = 3,859,230 m^3 yr^{-1}, "M-E" = 3,933,533 m^3 yr^{-1}). This was a result of the phosphorus concentration being substantially higher in "H-E" (annual flow-weighted mean = 91 mg m^{-3} versus 58 mg m^{-3} in "M-E"), probably due to agricultural land use and road construction in the former watershed. Algal bioassays suggest that approximately 42% of the total phosphorus loading through "M-E" was bioavailable, whereas approximately 56% of the "H-E" load was bioavailable. Most of the phosphorus was in the dissolved form, and the higher availability in stream "H-E" was a result of a higher dissolved load.

The observed atmospheric deposition rate of total phosphorus for the same water year was 9.0 mg m^{-2}. This figure is comprised of an overwinter deposition rate of 2.2 mg m^{-2} and a spring to fall deposition rate of 6.8 mg m^{-2}. This rate is much lower than the figure derived at Baptiste Lake during the 1978 water year (30.4 mg m^{-2}), which in itself was well below international values reported from the literature (Trew et al., 1981). Most of this phosphorus was in the dissolved form, but no data with respect to bioavailability were collected. Deposition rates will be measured again during 1981.

The fluctuations in the volume-weighted hypolimnetic total phosphorus concentrations of Ethel Lake are illustrated in Figure 5. Prior to total oxygen depletion (mid-June) the average concentration was 37 mg m^{-3}. Immediately thereafter a sharp increase in concentration was observed with a plateau being reached during July. A much smaller rate of increase commencing in early August coincides with the general increase in epilimnetic chlorophyll 'a', and may represent autochthonous inputs to the hypolimnion via sedimentation. An average concentration of 89 mg m^{-3} was determined for July and August; this suggests a net hypolimnetic increase of 52 mg m^{-3} or 204 kg.

At this time we are unable to determine whether this hypolimnetic increase represents an actual input to the lake via sediment release or simply a redistribution of the phosphorus mass within the water column. Many of the existing data regarding low external inputs during the May-June period and the vertical pattern of the hypolimnetic phosphorus gradient suggest that in fact that this represents a net input to the system. However, the fluctuations in epilimnetic phosphorus are such that a downward vertical transfer of phosphorus cannot be totally ruled out at this stage. An attempt will be made to derive a May-November phosphorus budget by using short-term (i.e., fortnightly) mass balancing after the acquisition of a second year of data, and hopefully this exercise will clarify the origins of this phosphorus pool. For the present discussion, it will be treated as an individual source.

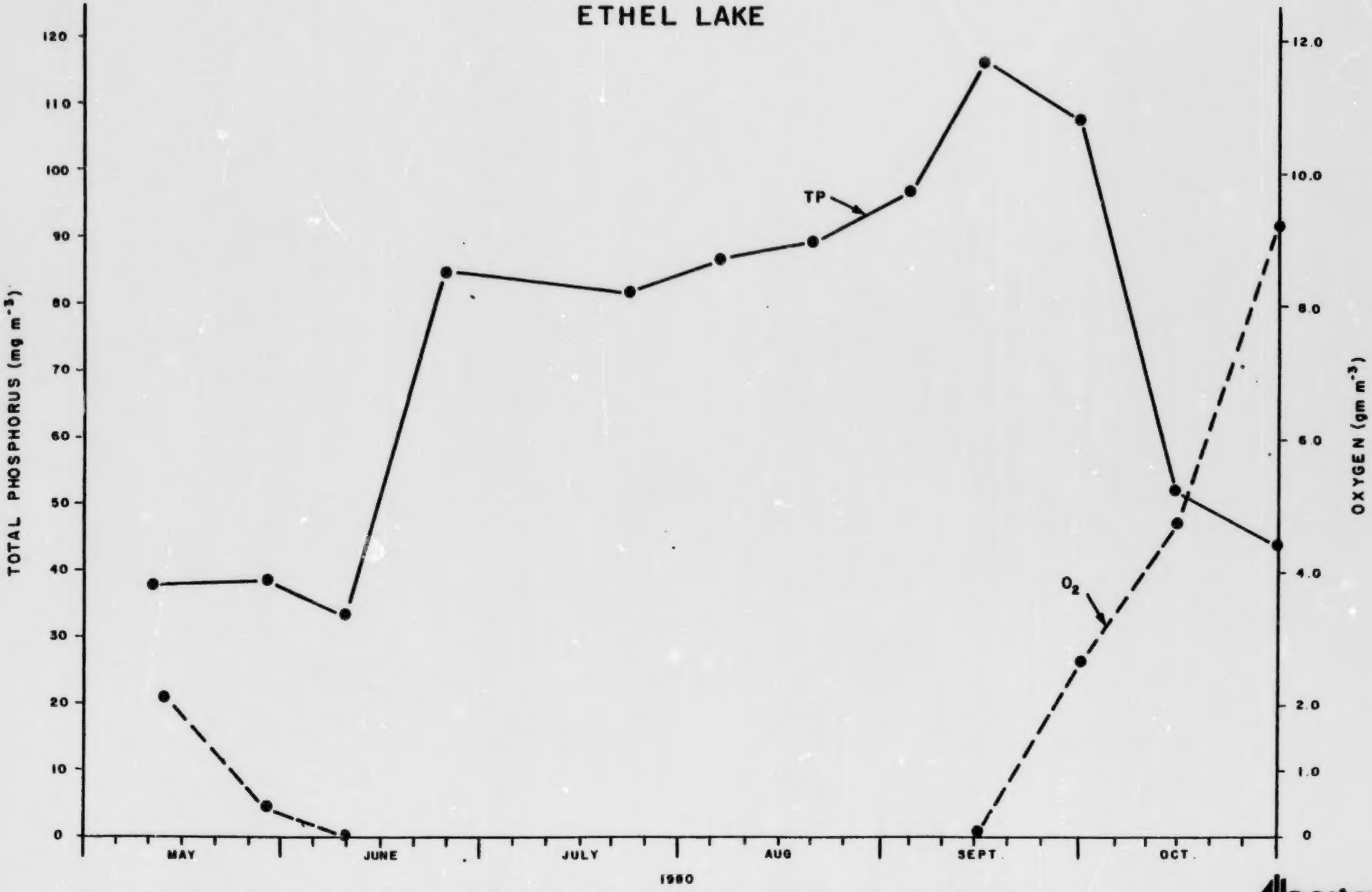

FIGURE 5 VOLUME-WEIGHTED HYPOLIMNETIC TOTAL PHOSPHORUS

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The "observed" loadings are summarized in Table 12.

TABLE 12. TOTAL PHOSPHORUS LOADINGS TO ETHEL LAKE (kg yr^{-1})

Stream "H-E"	351
Stream "M-E"	229
Atmospheric Deposition	43
Diffuse Runoff	211
Domestic/Recreational	55
Urban	13
Sediment Release	204
TOTAL	1106

This figure (1106 kg yr^{-1}) is at the lower end of the loading range for Ethel Lake presented in Table 11. However, 1981 was a low precipitation year, and the observed stream/atmospheric deposition inputs may be lower than the long-term averages.

The annual surface water outputs of total phosphorus have also been determined for Ethel Lake; these pathways include the outlet stream (109 kg yr^{-1}) and the licenced water withdrawal by Esso Resources Canada (40 kg yr^{-1}). Based upon this data, and ignoring both sediment resuspension and subsurface seepage as vectors for the movement of phosphorus through the lake, an annual retention estimate of .87 can be derived for the year 1979-80. This is approximately .10 units higher than either of the calculated retention estimates.

Detailed phosphorus profiles at 2.5 m intervals were recorded before and after autumn mixing to determine the effects of re-oxygenation on the hypolimnetic phosphorus pool. The total water column content was determined by mathematical integration and by total lake content volume - weighting. Both techniques suggest a net decrease in phosphorus content as a result of mixing (Table 13).

TABLE 13. TOTAL PHOSPHORUS CHANGES AT AUTUMN MIXING

DATE	LAKE MASS (kg)	AREAL CONCENTRATION ($mg\ m^{-2}$)
SEPT. 30	1150	2.50
OCT. 15	885	1.67
OCT. 30	878	1.07

On a whole - lake basis, 272 kg of total phosphorus is lost during the month of October, and most of the hypolimnetic loss occurs during the two week interval from Sept. 30 to Oct. 15, 1980 in conjunction with the depression of thermal density stratification in the upper hypolimnion. One tentatively concludes that a large portion of the hypolimnetic phosphorus pool in Ethel Lake is returned to the sediments by chemical precipitation (Golterman, 1975) and has little impact on overall lake concentrations during fall mixing, however no information regarding trans-thermocline migration of phosphorus during the period of stratification is available.

TROPHIC STATUS

PHYTOPLANKTON STANDING CROP

The mean epilimnetic phytoplankton standing crops [B] for the open water season, expressed as chlorophyll 'a', are presented in Table 14. The standard deviation of each mean is shown in brackets.

TABLE 14. MEAN EPILIMNETIC CHLOROPHYLL 'a' (mg m^{-3})

	n	Chl 'a'
TUCKER	12	16.6 (8.2)
ETHEL	13	10.8 (4.0)
MAY	11	10.7 (5.2)
HILDA	11	9.9 (10.2)
MOORE	12	7.9 (3.7)
BOURQUE	12	7.3 (3.4)
MARIE	12	6.6 (3.8)
PRIMROSE NORTH	6	16.7 (7.0)
PRIMROSE SOUTH	7	9.8 (5.1)
COLD	7	3.6 (2.2)

Three seasonal patterns in chlorophyll concentrations were evident. The most productive lake, Tucker, followed a pattern similar to other eutrophic lakes. A strong spring increase was followed by a distinct early summer minimum, with the highest levels observed during July and September(Figure 6). Autumn values subsequently declined from the mid-summer peaks.

May Lake (Figure 7) was somewhat similar to Tucker in seasonality, although the concentrations observed were much lower. In addition to suggested spring and summer blooms, a small autumn upsurge was detected in this lake. The date at which isothermal conditions appeared is illustrated by a vertical arrow. All other lakes (Figures 8 to 12) displayed more subdued fluctuations through spring and summer; a general increase in all cases began in early August and culminated with the peak pigment concentrations in mid to late October. At present it is not known whether this pattern is correlated with other standing crop parameters, or merely reflects altered carbon: chlorophyll ratios associated with community succession and changes in the physical/nutritional environment. These relationships will be further explored when cell number/cell volume data are available.

A correlation was observed between mean epilimnetic phosphorus and mean epilimnetic chlorophyll for the open water season:

$$\mathrm{Log}\,[B] = 0.873\,\mathrm{Log}\,[P] - 0.095$$

$$r^2 = 0.60$$

However, the correlation improved somewhat when Cold and Primrose Lakes were dropped from the data set (these lakes were subjected to a reduced sampling frequency):

$$\mathrm{Log}\,[B] = 0.666\,\mathrm{Log}\,[P] - 0.041$$

$$r^2 = 0.74$$

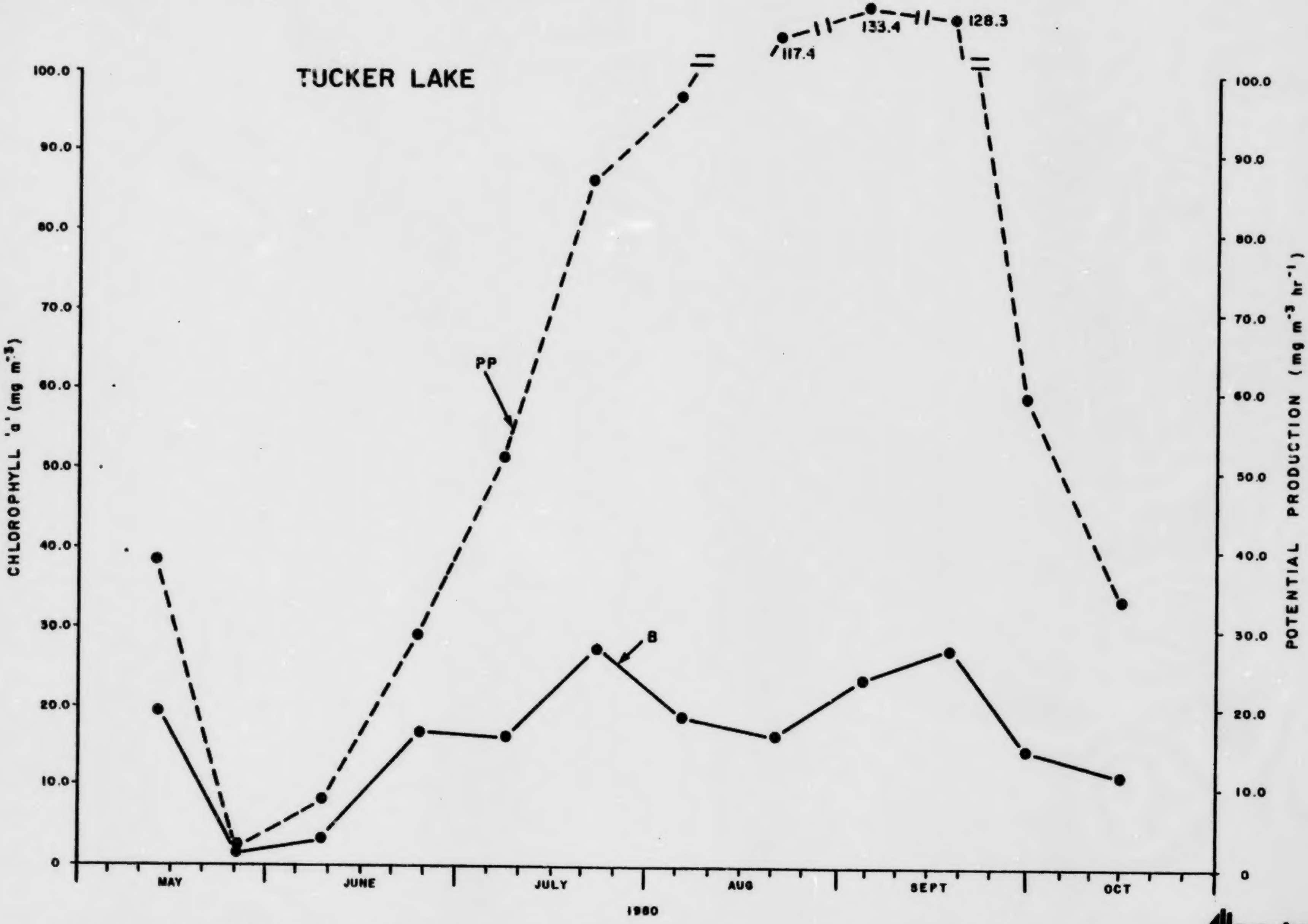

FIGURE 6 SEASONAL FLUCTUATIONS IN CHLOROPHYLL 'a' AND POTENTIAL PRODUCTION

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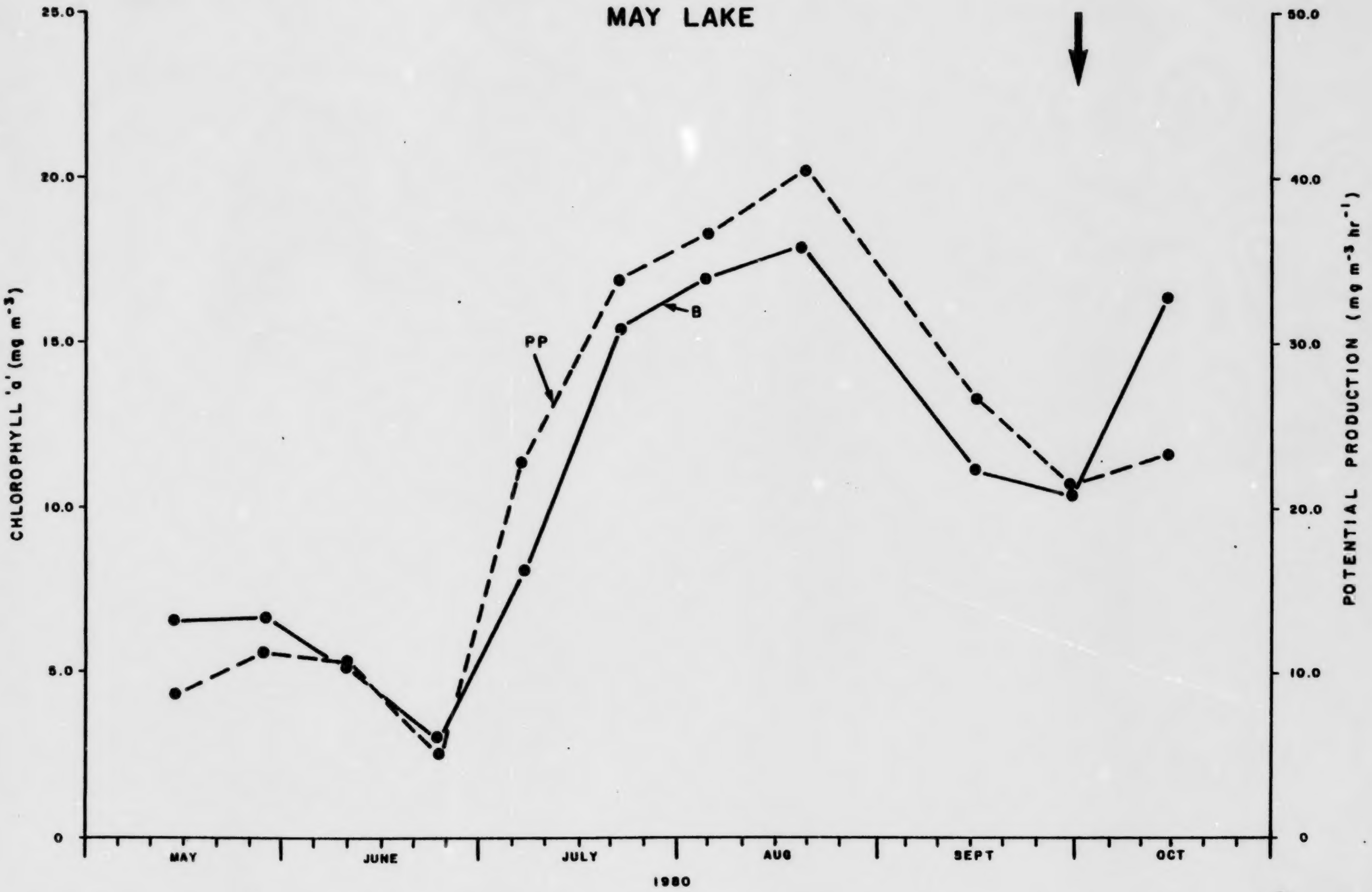

FIGURE 7 SEASONAL FLUCTUATIONS IN CHLOROPHYLL 'a' AND POTENTIAL PRODUCTION

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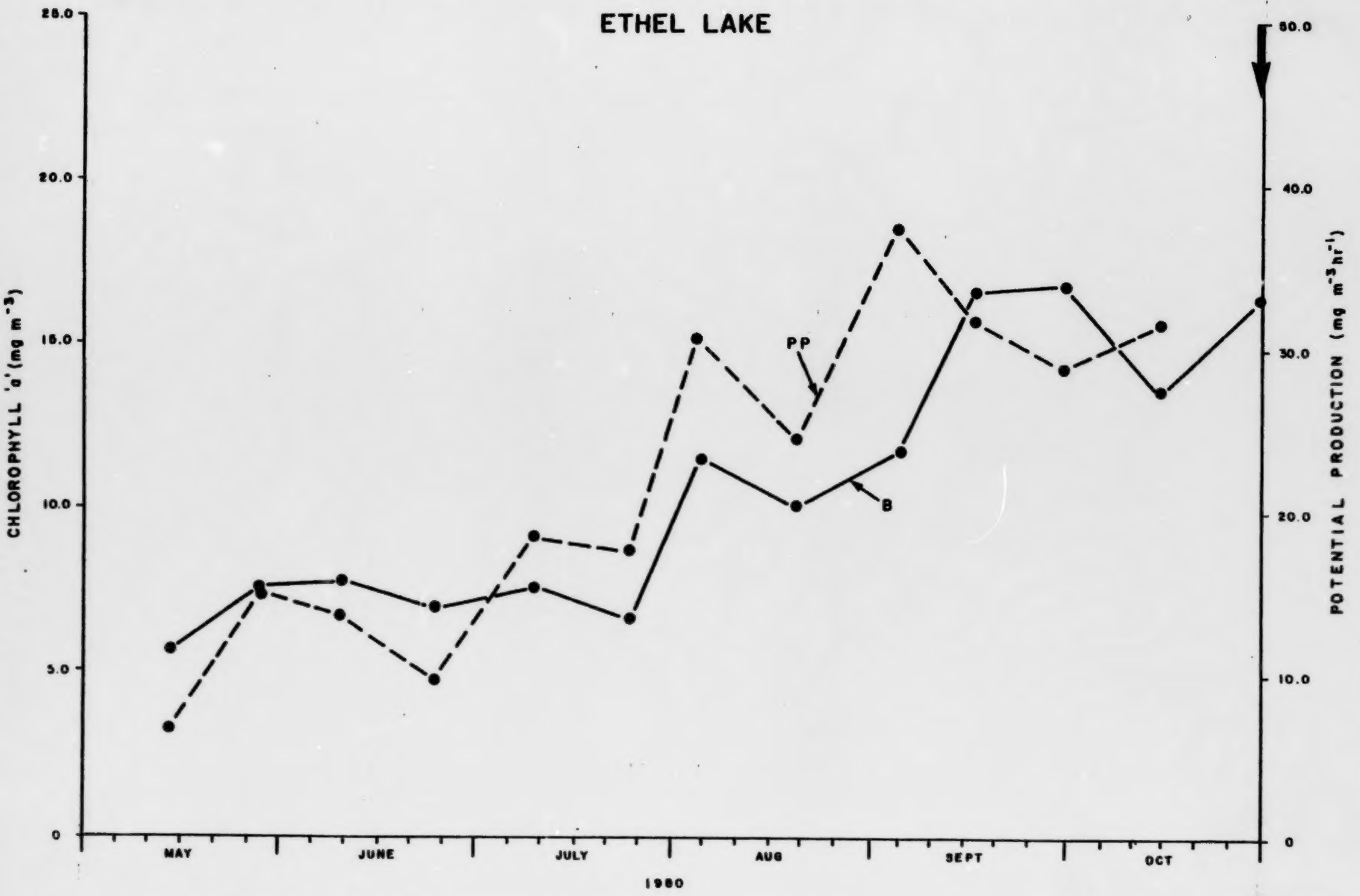

FIGURE 8 SEASONAL FLUCTUATIONS IN CHLOROPHYLL 'a' AND POTENTIAL PRODUCTION

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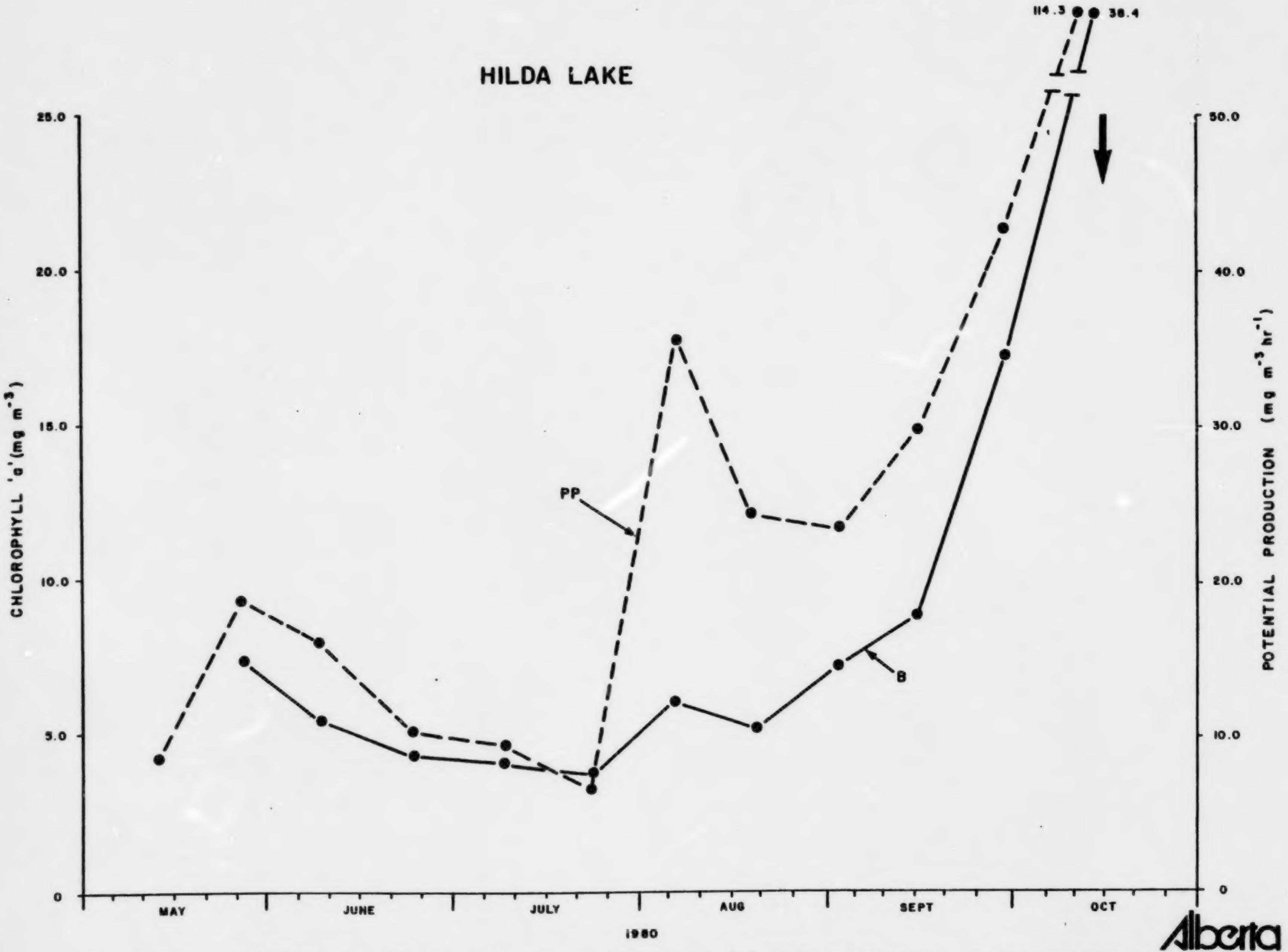

FIGURE 9 SEASONAL FLUCTUATIONS IN CHLOROPHYLL 'a' AND POTENTIAL PRODUCTION

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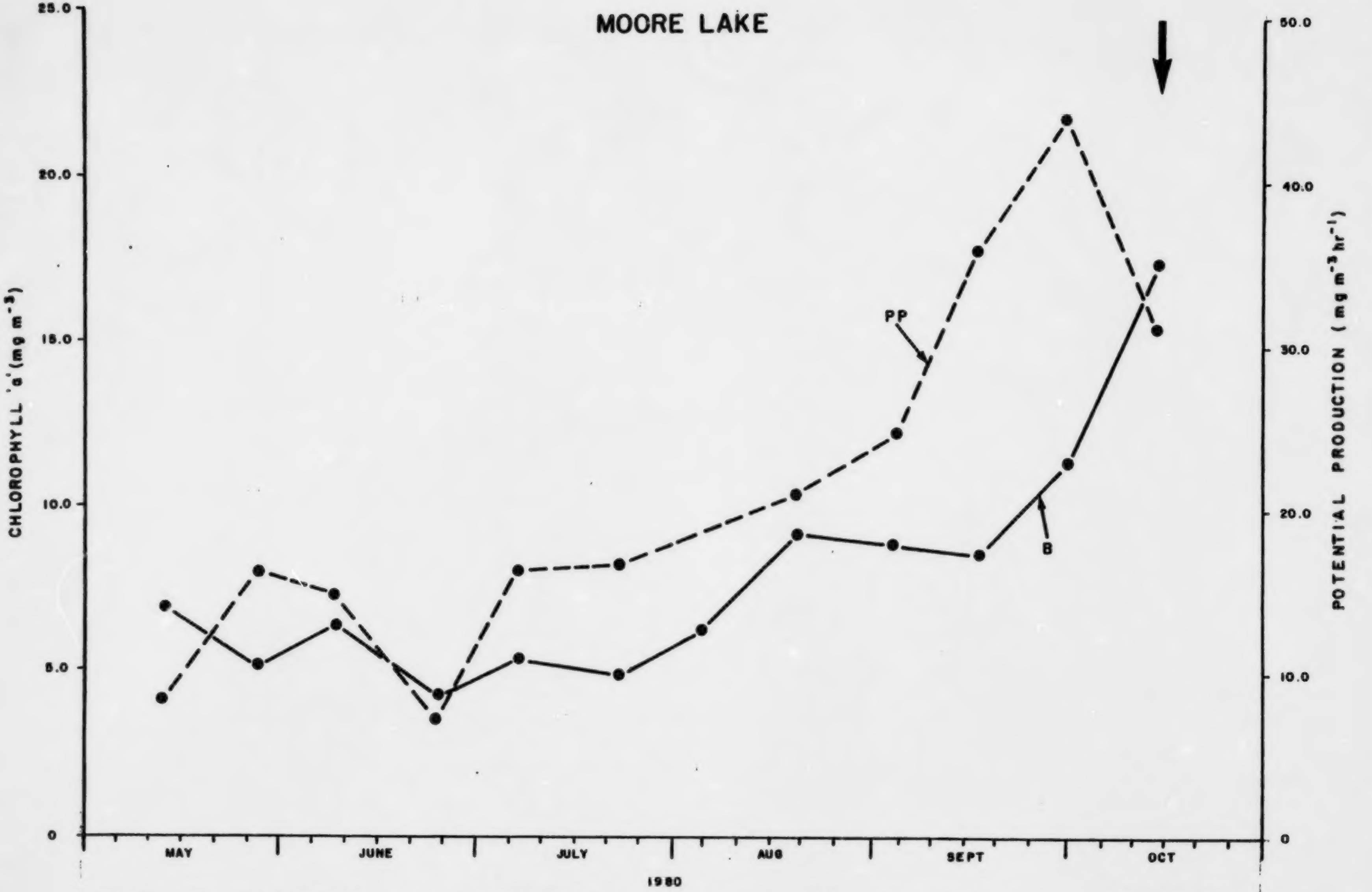

FIGURE 10 SEASONAL FLUCTUATIONS IN CHLOROPHYLL 'a' AND POTENTIAL PRODUCTION

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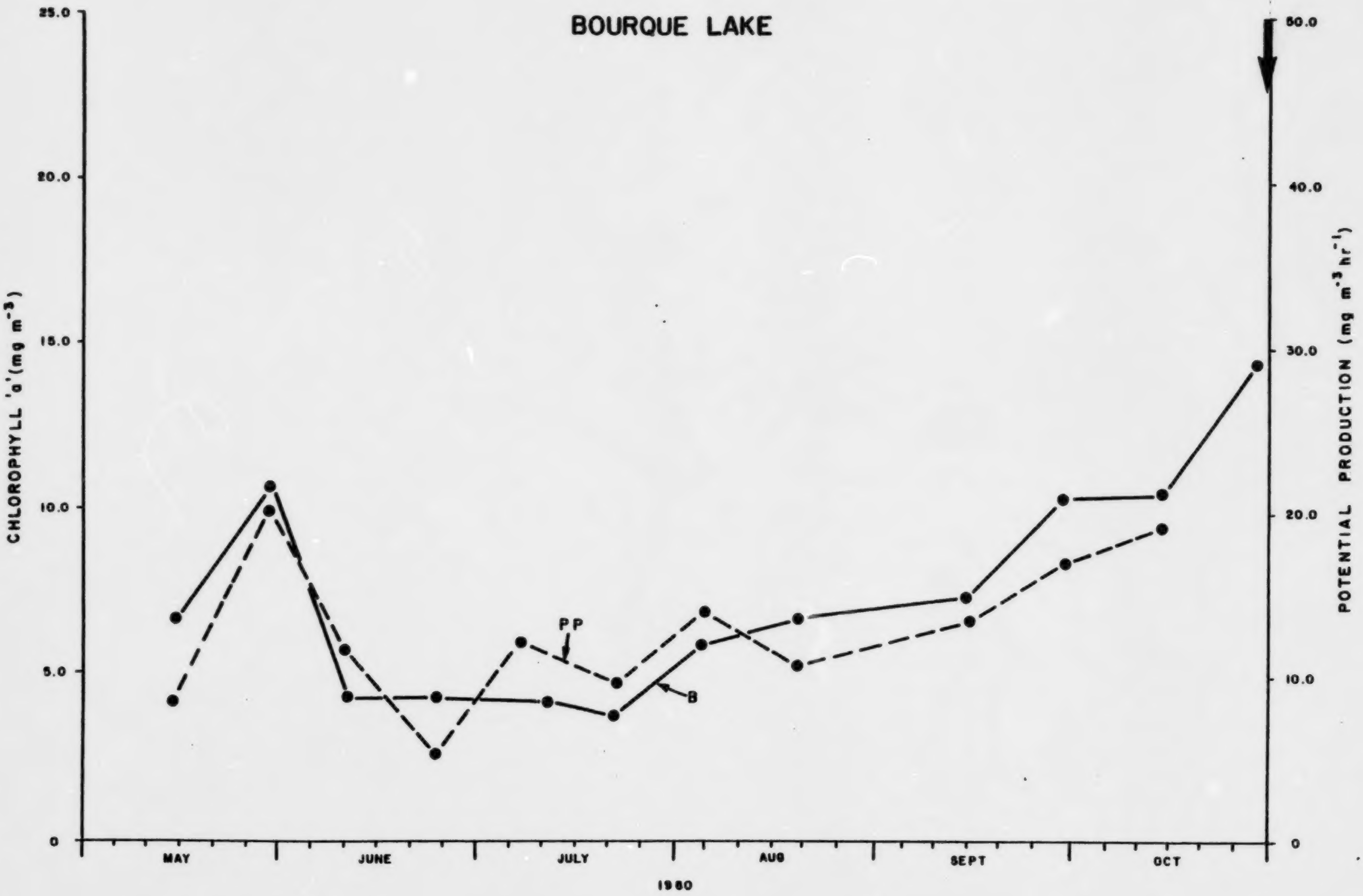

FIGURE II SEASONAL FLUCTUATIONS IN CHLOROPHYLL 'a' AND POTENTIAL PRODUCTION

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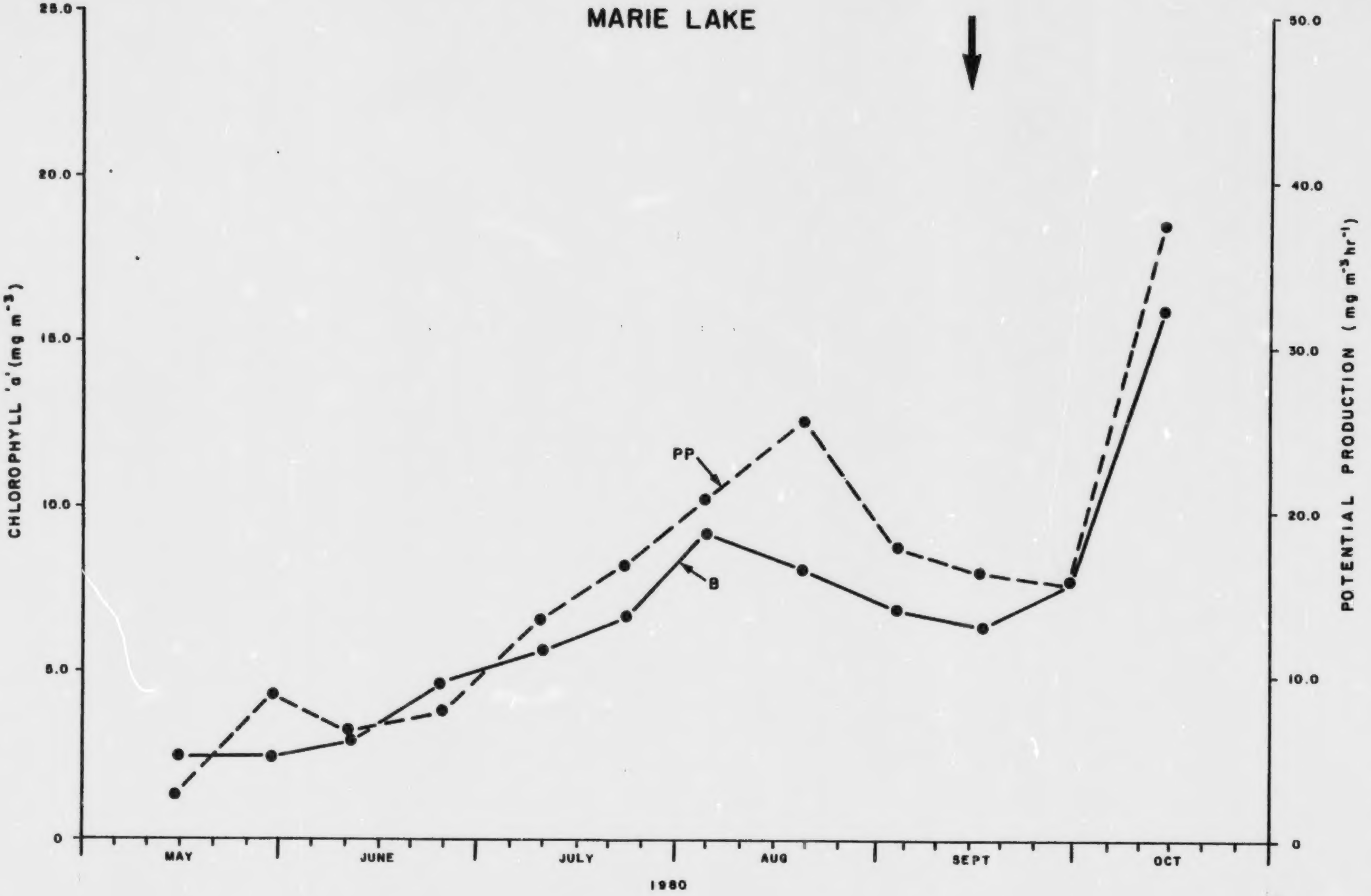

FIGURE 12 SEASONAL FLUCTUATIONS IN CHLOROPHYLL 'a' AND POTENTIAL PRODUCTION

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In situ profiles of chlorophyll 'a' were determined fortnightly in Ethel Lake at following depths: 0, 0.5, 1, 1.5, 2, 3, 6, 10, 15, 20, 25 m. These profiles are illustrated in Figures 13A - 13D. The depth of the euphotic zone, as defined by the point to which 1% of surface penetrating irradiance was observed, is illustrated by a horizontal arrow.

In most cases chlorophyll 'a' is observed below the euphotic zone, and only during late summer does one observe any pronounced vertical stratification. During spring and autumn the phytoplankton are generally homogenously distributed within the epilimnion.

The mean epilmnetic chlorophyll 'a' concentrations in Ethel Lake as determined by both the integrating sampler (at 20 sites each date), and by the mathematical integration of the in situ data (one site each date) are presented in Table 15.

IN SITU PRODUCTION (mg m^{-3} hr^{-1}) & CHLOROPHYLL (mg m^{-3})

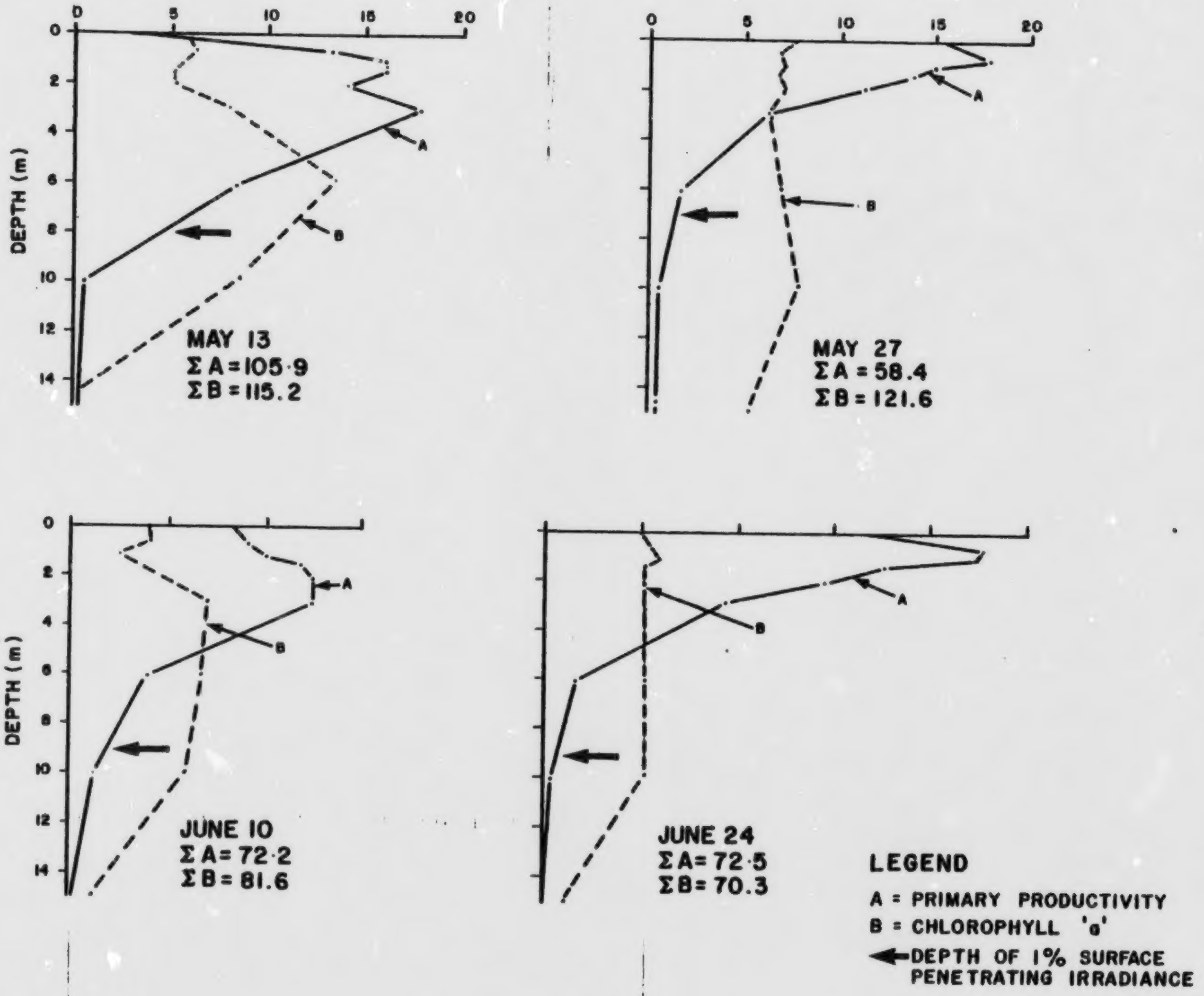

FIGURE 13 A

IN SITU PRODUCTION (mg m^{-3} hr^{-1}) & CHLOROPHYLL (mg m^{-3})

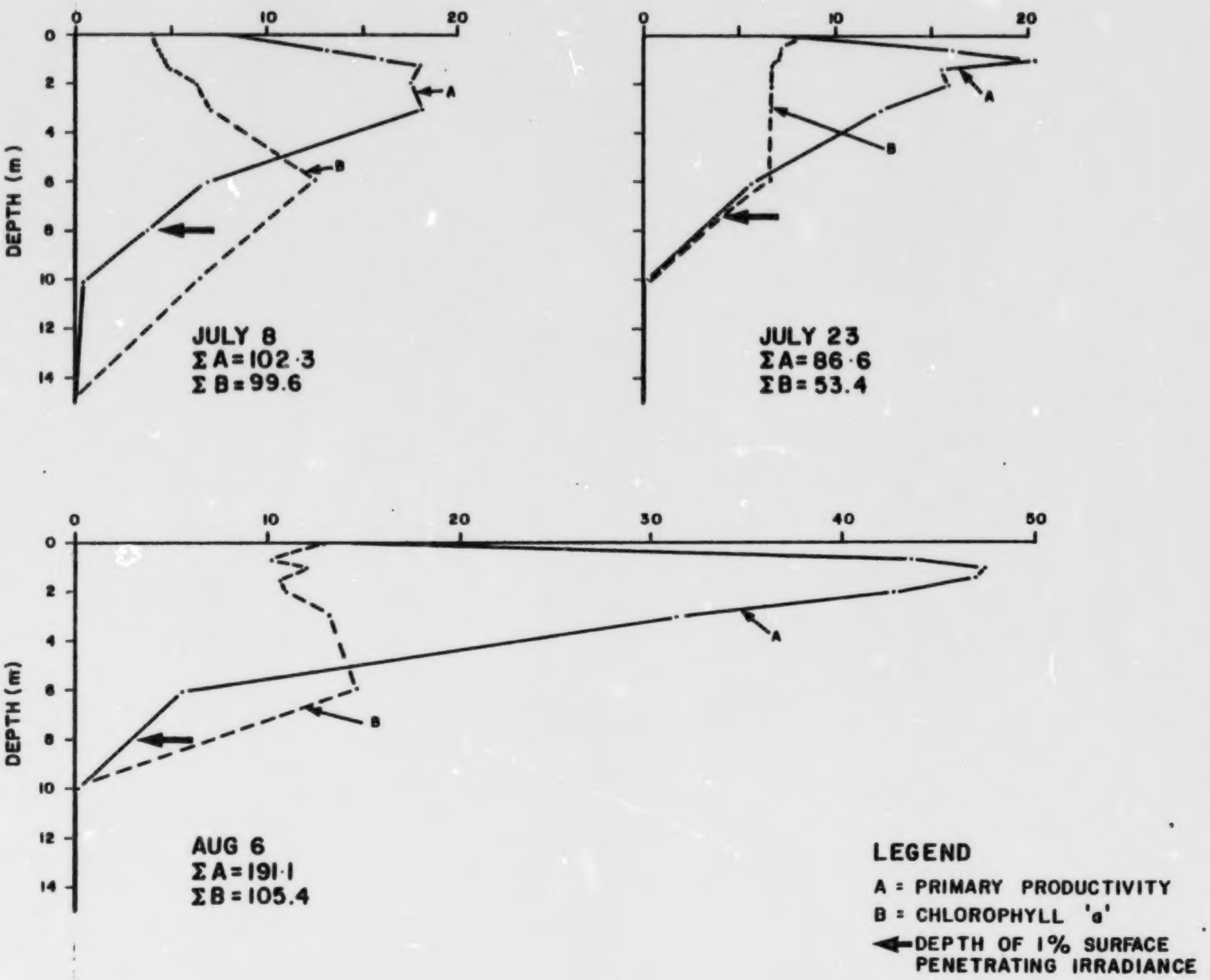

FIGURE 13 B

IN SITU PRODUCTION (mg m^{-3} hr^{-1}) & CHLOROPHYLL (mg m^{-3})

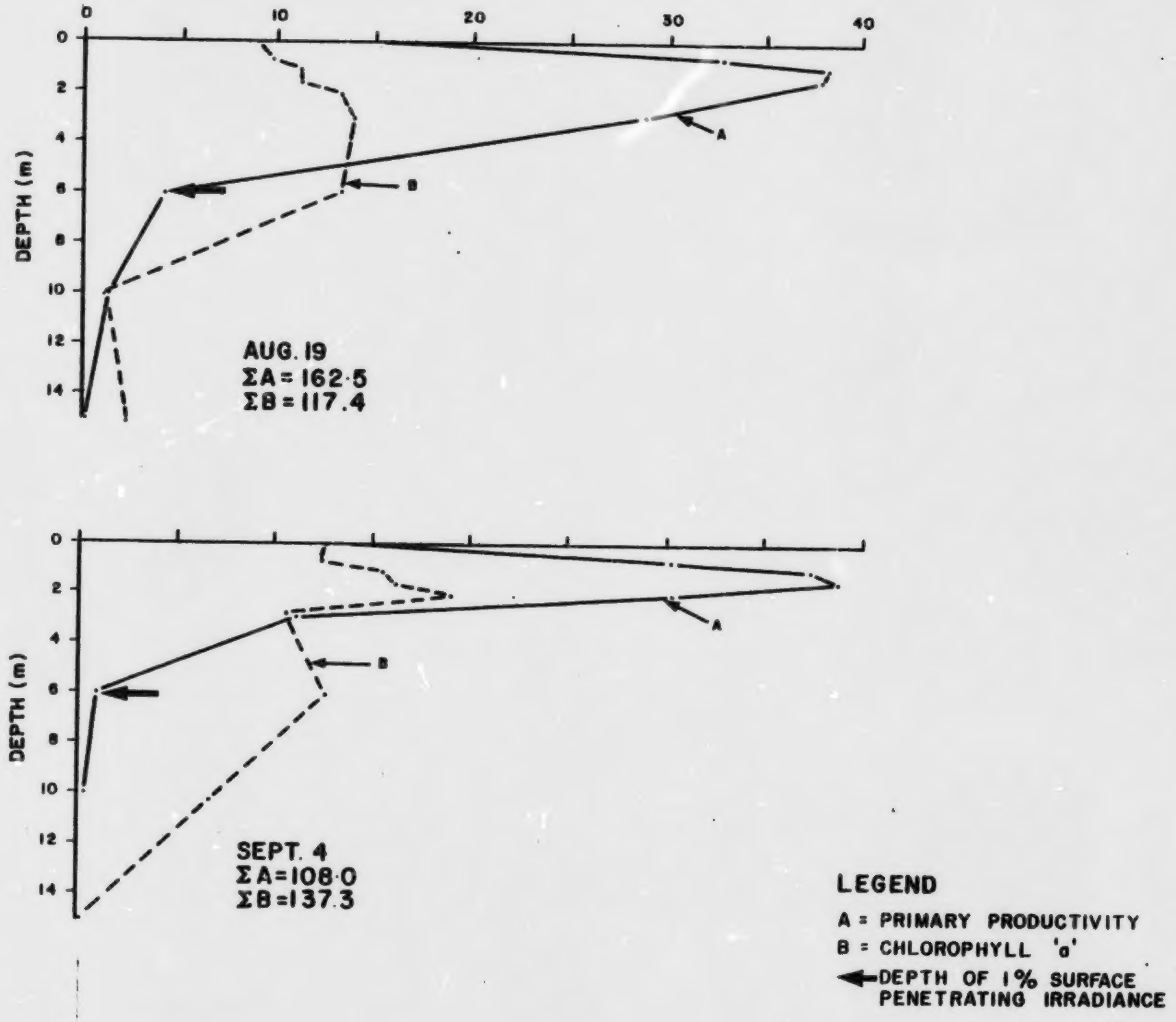

FIGURE 13 C

IN SITU PRODUCTION (mg m^{-3} hr^{-1}) & CHLOROPHYLL (mg m^{-3})

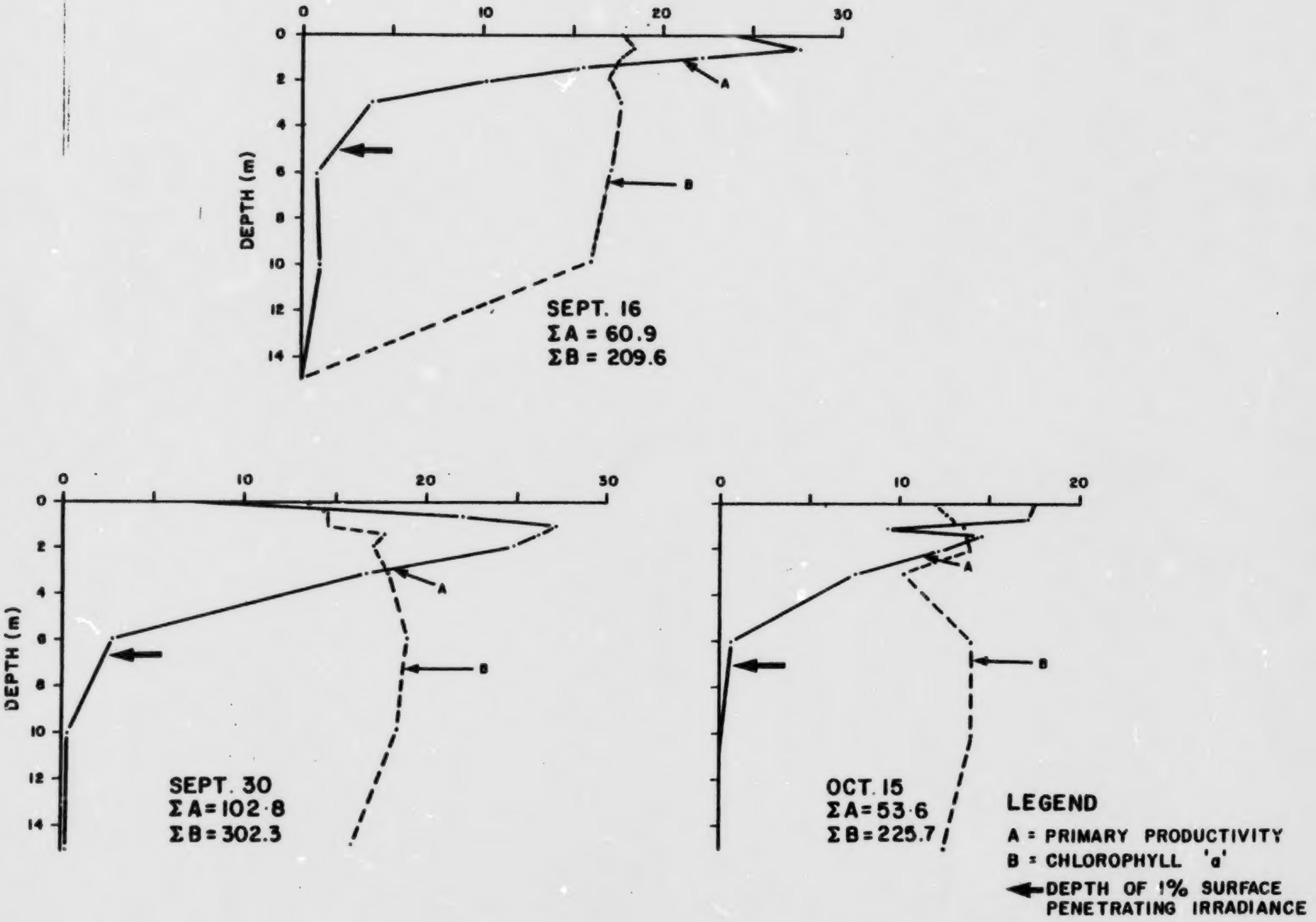

FIGURE 13 D

TABLE 15. MEAN DAILY CHLOROPHYLL 'a' CONCENTRATIONS (mg m^{-3})

DATE	IN SITU	INTEGRATING SAMPLER
May 13	8.32	5.62
May 27	6.86	7.63
June 10	5.73	7.74
June 24	5.08	7.00
July 8	7.67	7.53
July 23	6.71	6.73
Aug. 5	12.69	11.56
Aug. 18	12.43	10.07
Sept. 4	11.92	11.88
Sept. 16	17.02	16.70
Sept. 30	17.53	16.92
Oct. 15	12.99	13.60

This comparison is somewhat gross, since the in situ figures had to be derived from integrals corresponding to the sample depth nearest to thermocline which in some cases may have meant a differential of up to 2 meters from the depth of the integrating sampler. In addition, the in situ figures were based upon samples from a single location whereas the other set were composites from the whole lake. In spite of these problems however, a close correlation was observed between these two estimates:

$$\mathrm{LOG}[B_{IS}] = 1.013\ \mathrm{LOG}[B_{INT}] - 0.013$$

$$r^2 = 0.79$$

The reason for this may be partially due to the spatial distribution of the phytoplankton, which in less productive lakes may be uniform.

CHLOROPHYLL DISTRIBUTION SURVEYS

In order to provide a quantitative frame of reference for the design of the sampling program, chlorophyll surveys similar to those conducted on Baptiste Lake during 1976 were carried out on Ethel, Marie, Hilda, and Tucker Lakes. On each survey date approximately 40-50 separate samples were taken from an arbitrary grid across the lake. Although the sampling approaches were different (surface grabs at Baptiste versus integrated samples in the 1980 work) the data have been tentatively combined to illustrate the potential relationships between sample statistics. The data base will be enlarged during 1981 to include more integrated sample surveys from productive lakes.

The sample variances (s^2), means ($\bar{x}$), $s^2:\bar{x}$ ratios, and distribution types are presented in Table 16.

TABLE 16. CHLOROPHYLL SURVEY STATISTICS

LAKE	DATE	s^2	$\bar{x}$	$s^2:\bar{x}$	DISTRIBUTION
HILDA	22-7-80	0.14	3.6	0.04	REGULAR
ETHEL	23-7-80	0.56	6.9	0.08	REGULAR
MARIE	20-8-80	0.62	7.3	0.08	REGULAR
TUCKER	3-9-80	41.56	18.4	2.26	CONTAGIOUS
BAPTISTE	31-5-76	11.64	12.7	0.92	POISSON
BAPTISTE	16-6-76	87.96	22.2	3.96	CONTAGIOUS
BAPTISTE	5-7-76	1091.54	71.1	15.35	CONTAGIOUS
BAPTISTE	15-8-76	145.59	68.1	2.14	CONTAGIOUS
BAPTISTE	7-9-76	651.83	50.11	13.01	CONTAGIOUS

According to Elliot (1977) an $s^2:\bar{x}$ ratio >1 is indicative of a contagious, or clumped distribution whereas an $s^2:\bar{x}$ ratio <1 is indicative of a regular, or even distribution. Thus, the phytoplankton of the less productive lakes appear to be more evenly distributed, whereas in the more productive waters of Tucker and Baptiste the phytoplankton are clumped, perhaps necessitating alternate sampling strategies.

The limited data available here suggest a progression from uniform → random → clumped distribution as chlorophyll concentrations increase through the range of 7.3 to 18.4 mg m^{-3}. This generally corresponds with the standing crop levels which have been historically used to separate mesotrophic from eutrophic lakes (10 - 15 mg m^{-3}); consequently, there may exist a taxonomic correlation as well as a density basis for distribution patterns at higher standing crops, since many Cyanophyceae tend to clump.

Furthermore, these data suggest that the distribution in any one lake can evolve from regular to contagious during the course of the same growing season as the mean fluctuates. In order that a statistically consistent sampling approach is utilized on any given day, the data suggest that the number of samples could vary according to the mean phytoplankton standing crop present. This in turn presents something of a dilemma since one cannot necessarily anticipate the standing crop prior to the sampling event. A reasonable solution would appear to involve taking an arbitarily large number of samples on all days which was the strategy during 1980 (20 samples).

A regression analysis suggests that the variance of the above data is strongly dependent upon the mean (Fig. 14):

$$\text{LOG [VARIANCE]} = 2.91 \ \text{LOG [MEAN]} - 2.42$$

$$r^2 = 0.92$$

Theoretically, one could utilize such a relationship to determine the minimum sample size required to produce an estimate of the population mean within a desired standard error (provided that some idea of the mean is already available):

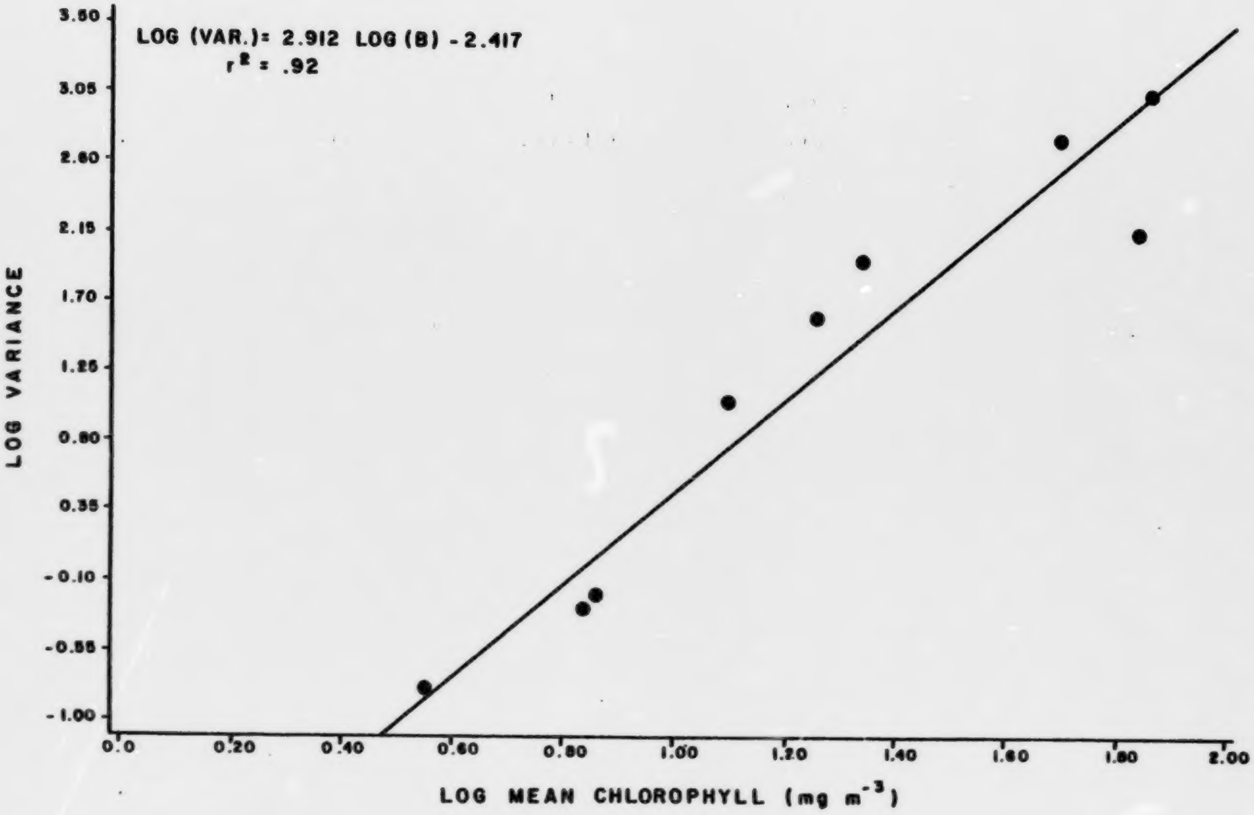

FIGURE 14 THE VARIANCE AS A FUNCTION OF THE MEAN IN CHLOROPHYLL DISTRIBUTION SURVEYS

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$$n = \frac{s^2}{D^2 \cdot \bar{x}^2} \quad \text{(Regular distribution)}$$

or

$$n = \frac{1}{D^2} \quad \frac{1}{\bar{x}} + \frac{1}{k} \quad \text{(Contagious distribution)}$$

where D = ratio of the standard error to the arithmetic mean

$\bar{x}$ = sample mean

k = exponent of the negative binomial given by Elliot (1977) as:

$$k = \frac{x^2}{s^2 - \bar{x}}$$

Using Ethel Lake (23-7-80) as an example (Table 16), the number of samples required to produce an estimate of the mean with a ±10% standard error would be:

$$n = \frac{0.56}{(.10)^2\ (6.9)^2}$$

$$= 1$$

The highest standing crop observed in any lake during 1980 was 38.4 mg m^{-3} in Hilda Lake (October 15). The variance associated with this mean could be estimated from the previous regression as 155.0, and using the formula for clumped distributions indicates that 16 samples would be required to fall within a standard error of ±10%. This analysis suggests that the sampling program conducted during 1980 may have been adequate to determine representative whole-lake standing crop estimates.

In developing the concept of the theoretical upper limit to euphotic zone chlorophyll content (ΣB_{eu}), Talling (1965) assumed that at high chlorophyll concentrations extinction was essentially due to algal material. The chlorophyll content of the euphotic zone in which the phytoplankton are homogenously distributed is given by:

$$\Sigma B_{eu} = B \times z_{eu}$$

or

$$\Sigma B_{eu} = B \times \frac{3.7}{\varepsilon_{min}}$$

At the highest chlorophyll concentration:

$$\varepsilon\ min = \varepsilon_s B$$

therefore, $$\Sigma B_{max} = \frac{3.7}{\varepsilon_s}$$

Since ε_s has already been defined on Ethel Lake as .027, the theoretical upper limit of euphotic chlorophyll content for this lake could be estimated at 137 mg m^{-2}. If the product $\varepsilon \cdot z_{eu}$ is equivalent to 4.4 in these lakes as discussed previously, then the theoretical upper limit could be estimated at 163 mg m^{-2}. However, the maximum figure for ΣB_{eu} observed on Ethel lake was approximately 103 mg m^{-2} (Sept. 30, 1980), distributed over a euphotic zone of 6.0 m depth. This analysis suggests that the lake is well below its theoretical upper limit.

A preliminary set of observations was performed on the outlet stream of Ethel Lake to test out the feasibility of using such a location to determine in-lake characteristics. Daily chlorophyll observations were made from May 5 to October 27, 1980; when daily discharge data becomes available a detailed flow-weighted mean will be calculated. For the interim, a flow-weighted mean based upon miscellaneous discharge measurements (i.e., fortnightly) during the period of in-lake observations (May 14, 1980, to October 31, 1980) has been determined as 5.0 mg m^{-3}. The discrepancy between this value and the in-lake observation (10.8 mg m^{-3}) may be partially due to the influence of the extensive macrophyte beds in the shallow bay on the east side of the lake. Competition for nutrients, coupled with an extremely low water throughput during 1980 may have reduced the growth and flushing of the phytoplankton community in that part of the lake.

PHYTOPLANKTON PRIMARY PRODUCTION

Primary production measurements using the radiocarbon method of Steemann-Nielsen (1952) were performed both in situ and under controlled light and temperature; the latter are hitherto referred to as potential production. The light level at the depth of incubation was held constant at approximately 360 foot-candles (30 watt cool-white) while the temperature was varied according to ambient. Samples for all lakes were incubated under identical conditions during each field trip, and 10 μC of $NaH^{14}CO_3$ were used as the innoculum. The use of the incubator has removed the influence of day to day climatic variation on the photosynthetic measurements, and allows one to more clearly perceive the factors controlling primary production through between-lake comparisons.

The mean, epilimetic potential production values [PP] during the open-water season are presented in Table 17, with the standard deviations in brackets.

TABLE 17. MEAN EPILIMNETIC POTENTIAL PRODUCTION (mg C m^{-3} hr^{-1})

	n	PP
Tucker	12	65.4 (45.9)
May	11	21.8 (11.9)
Hilda	12	26.4 (29.3)
Ethel	12	22.0 (10.0)
Moore	12	21.7 (11.1)
Marie	12	15.8 (9.3)
Bourque	11	12.5 (4.4)
Primrose North	8	55.7 (29.6)
Primrose South	8	17.0 (4.9)
Cold	8	6.4 (2.3)

The potential production values exhibited a strong seasonal correlation with chlorophyll concentrations in most lakes. However, some of the less productive lakes such as Bourque and Moore, which displayed a limited range, had weaker statistical correlations. The individual regression equations are summarized in Table 18.

TABLE 18. POTENTIAL PRODUCTION - CHLOROPHYLL REGRESSIONS

MARIE	$\mathrm{LOG}\ [PP] = 1.098\ \mathrm{LOG}\ [B] + 0.286$	$r^2 = .84$
TUCKER	$\mathrm{LOG}\ [PP] = 1.373\ \mathrm{LOG}\ [B] + 0.083$	$r^2 = .90$
MAY	$\mathrm{LOG}\ [PP] = 1.097\ \mathrm{LOG}\ ]B] + 0.199$	$r^2 = .89$
HILDA	$\mathrm{LOG}\ [PP] = 1.034\ \mathrm{LOG}\ [B] + 0.439$	$r^2 = .82$
ETHEL	$\mathrm{LOG}\ [PP] = 1.264\ \mathrm{LOG}\ [B] + 0.049$	$r^2 = .75$
BOURQUE	$\mathrm{LOG}\ [PP] = 0.636\ \mathrm{LOG}\ [B] + 0.579$	$r^2 = .49$
MOORE	$\mathrm{LOG}\ [PP] = 0.934\ \mathrm{LOG}\ [B] + 0.478$	$r^2 = .48$

When daily data from all lakes was combined, a good correlation was again observed (Fig. 15).

$$\mathrm{LOG}\ [PP] = 1.163\ \mathrm{LOG}\ [B] + 0.231$$

$$r^2 = 0.79$$

A strong correlation was found between mean potential production with both mean total phosphorus and mean total biologically available phosphorus during the open-water season.

$$\mathrm{LOG}\ [PP] = 1.501\ \mathrm{LOG}\ [P] + 3.563$$

$$r^2 = 0.82$$

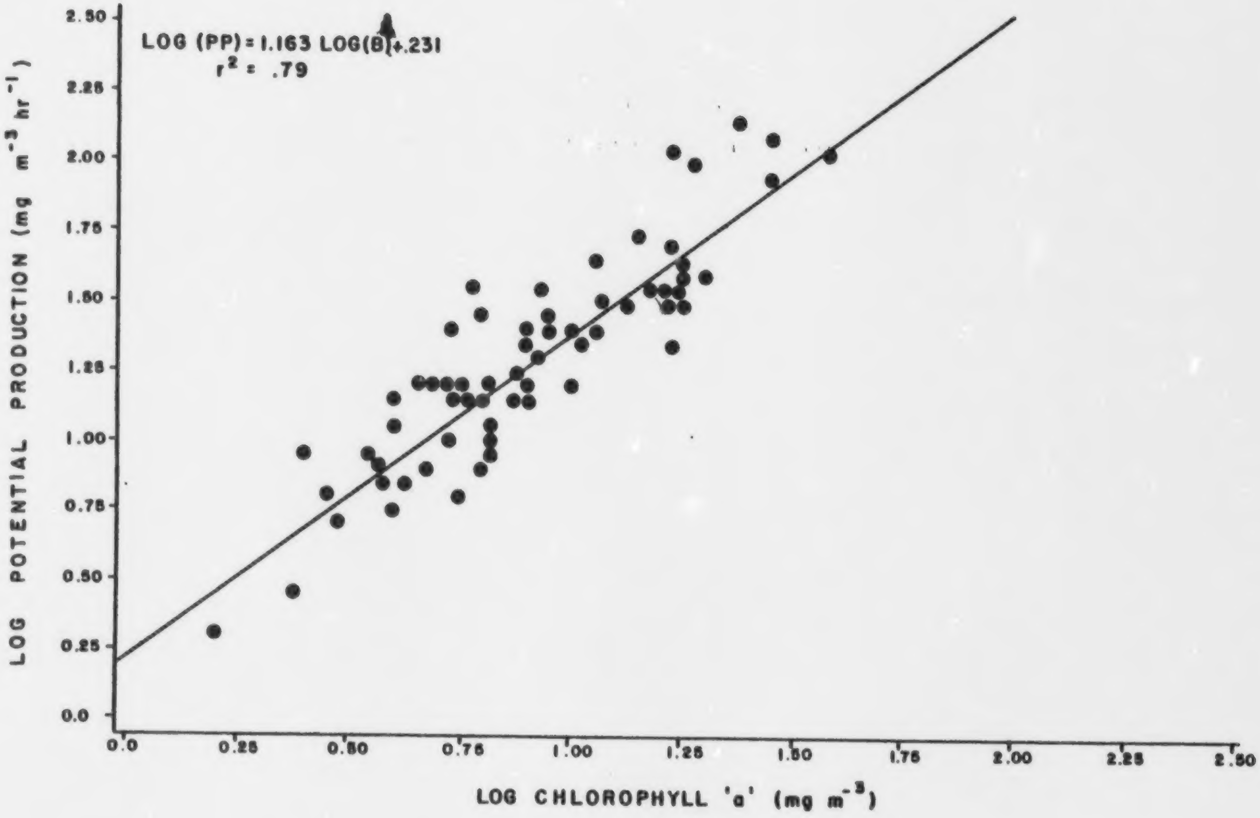

FIGURE 15 POTENTIAL PRODUCTION AS A FUNCTION OF CHLOROPHYLL 'a' IN THE STUDY LAKES

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$$\text{LOG [PP]} = 1.086 \text{ LOG [TBAP]} + 3.644$$
$$r^2 = .76$$

A photosynthetic index (PHI-carbon fixation per unit chlorophyll 'a') was calculated for each lake on each sampling day using the potential production data. Generally, the highest photosynthetic indices were observed in July and August, even though the peak chlorophyll values occurred in autumn in the mesotrophic lakes.

The mean photosynthetic indices for each lake are presented in Table 19.

TABLE 19. MEAN PHOTOSYNTHETIC INDEX

	n	PHI
TUCKER	12	3.57
HILDA	11	3.11
MOORE	12	2.69
MARIE	12	2.37
ETHEL	12	2.11
MAY	11	2.01
BOURQUE	11	1.99
PRIMROSE NORTH	8	2.98
PRIMROSE SOUTH	8	2.09
COLD	8	2.12

A correlation was found between the mean photosynthetic index and the mean total phosphorus concentration (Fig. 16).

$$\text{[PHI]} = 2.531 \text{ LOG [P]} + 6.247$$
$$r^2 = .69$$

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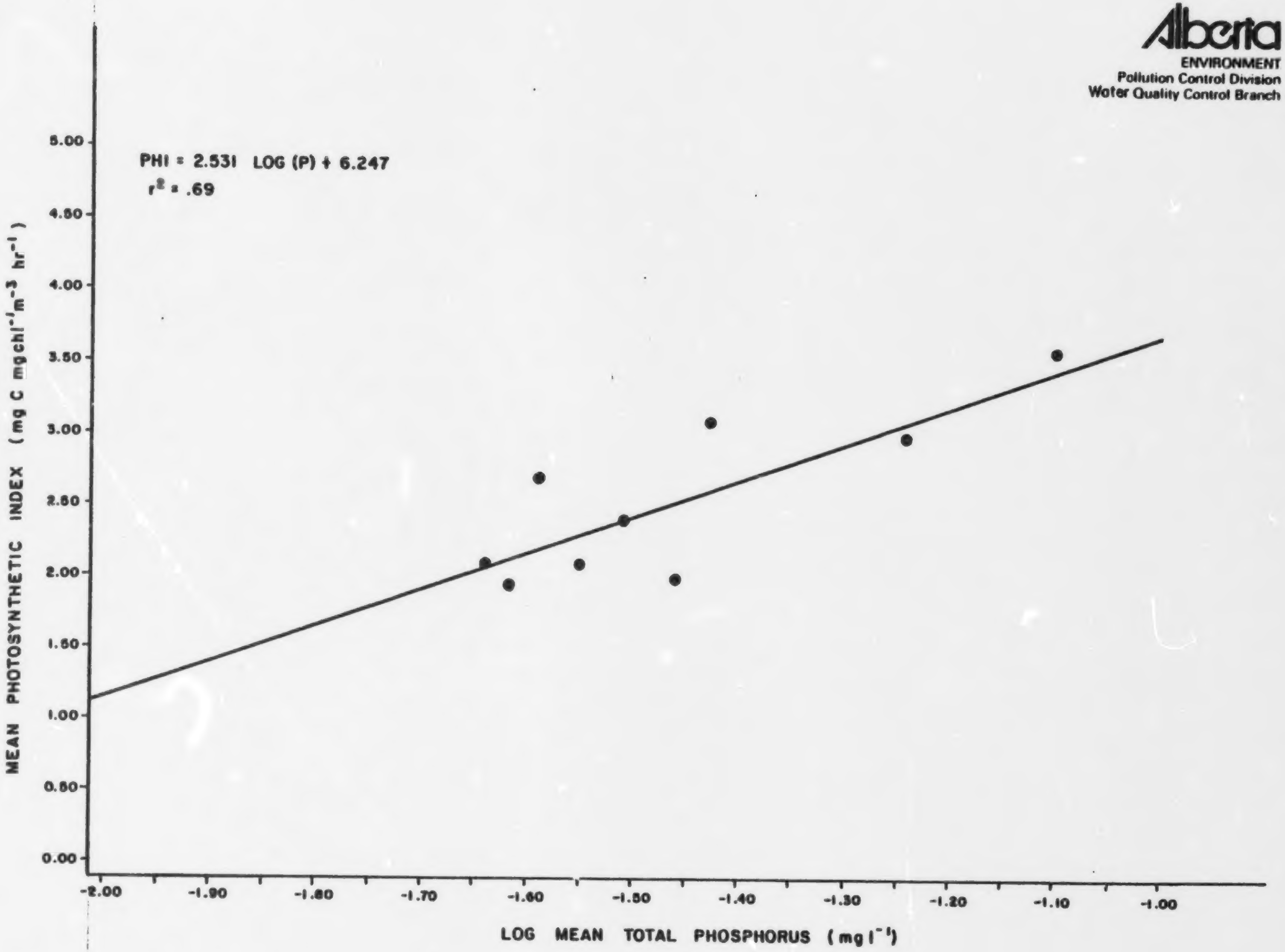

FIGURE 16 MEAN PHOTOSYNTHETIC INDEX AS A FUNCTION OF MEAN TOTAL PHOSPHORUS

When this index was compared against total biologically available phosphorus a weak correlation was observed:

$$[PHI] = 1.016 \text{ LOG } [TBAP] + 4.571$$
$$r^2 = 0.29$$

A relationship to nitrogen concentration was also suggested by performing similar analyses:

$$[PHI] = 2.901 \text{ LOG } [TN] + 2.468$$
$$r^2 = 0.64$$
$$[PHI] = 2.258 \text{ LOG } [TIN] + 5.672$$
$$r^2 = 0.73$$

The dependence of the photosynthetic index on the composition of the nutrional environment will be explored with multiple regression when the data set has been expanded.

In situ primary production measurements were made in Ethel Lake at the same depths that were used to define chlorophyll profiles (Figure 13A - 13D). The mean epilimnetic hourly fixation rate observed in situ [A] was 11.9 mg C m,$^{-3}$ whereas the incubator average was 22.0 mg C m^{-3}. We point out, however, that the incubator is a totally artificial environment in terms of light quality and quantity, and is not intended to simulate actual water column conditions.

On most sampling days, light levels and carbon fixation in the incubator were well below saturation, which usually occurred in the range of 500-2500 foot-candles in the lake. However, on the last three sampling trips of Sept. 16, 30, and Oct. 15, 1980 carbon fixation in the incubator was higher than the maximum rates observed in situ. This was probably due to the effects of diminishing light levels at that time of year.

Unlike the comparisons between potential production and chlorophyll, there was a very low slope and no correlation ($r^2 = .00$) between epilimnetic integral primary production and epilimnetic integral chlorophyll. This is probably the result of several incubations being conducted on overcast days, and also due to the fact that the late season chlorophyll peaks coincided with low ambient light. This lack of relationship is not always the case with in situ work, however, as moderately strong correlations were found on Baptiste Lake when various temporal/spatial expressions of both parameters were compared (Trew et al., 1981). However, more observations were available on Baptiste, and the chlorophyll pattern in that eutrophic lake followed the seasonal irradiance pattern more closely.

The mean, maximum rate of photosynthesis (A_{max}) in Ethel lake was 25.5 mg C m^{-3} hr^{-1}; this is much lower than the figure observed in Baptiste Lake (80.0 mg C m^{-3} hr^{-1}) when calculated upon a similar sampling regime. Once again, there was a weak correlation between A and B at the depth of A_{max} ($r^2 = .42$), probably due to the small number of observations as well as the wide range of climatic conditions under which the observations were made.

The mean daily rate of integral photosynthesis was determined over the period of observation (155 days) to be 918 mg m^{-2} day^{-1}. However, when this figure is extrapolated to a full year and assuming the contribution under ice is relatively small a figure of approximately 400 mg C m^{-2} day^{-1} could be derived. The annual figure derived for Baptiste Lake was 846 mg (m^{-2} day^{-1}).

ASH-FREE DRY WEIGHT

Determinations of ash-free dry weight, or particulate organic seston, were made on a daily basis on all lakes. Theoretically, this parameter is non-specific to the primary producers since any particulate material retained by a glass-fibre filter will contribute to the observed concentration (i.e. phytoplankton, zooplankton, detritus, etc.).

Correlation analyses were performed with all daily chlorophyll, potential production, and algal growth potential data and in each case very weak relationships were found:

$$\text{Log [AFDW]} = 0.686 \text{ Log [B]} + 2.582 \qquad r^2 = .31$$

$$\text{Log [AFDW]} = 0.668 \text{ Log [PP]} + 2.333 \qquad r^2 = .45$$

$$\text{Log [AFDW]} = 0.324 \text{ Log [AGP]} + 2.695 \qquad r^2 = .16$$

Consequently, routine measurement of this parameter is being discontinued in 1981.

ALGAL GROWTH POTENTIAL

The mean total algal growth potential (AGP) data for seven lakes as determined with the Selenastrum sp. assay (Hamilton et al., 1981) are presented in Table 22.

With the exception of Moore Lake, which showed no discernible pattern, the mesotrophic water bodies generally displayed their highest total algal growth potentials during September and October coincident with the peaks in standing crop/potential production as discussed previously. By contrast, Tucker Lake was high during both spring and autumn.

Table 20. MEAN TOTAL ALGAL GROWTH POTENTIAL (CELLS ML^{-1})

LAKE	AGP
TUCKER	141
HILDA	63
ETHEL	54
MAY	46
MOORE	45
MARIE	41
BOURQUE	28

On a daily basis (incorporating all observations from all lakes), algal growth potential was not as strongly correlated with potential production or chlorophyll as were the latter two parameters with each other, i.e.

$$\text{Log [PP]} = 0.591 \text{ Log [AGP]} + 0.348 \qquad r^2 = 0.40$$

$$\text{Log [B]} = 0.460 \text{ Log [AGP]} + 0.177 \qquad r^2 = 0.44$$

These weaker relationships may be partially explainable by the in-lake distribution of nutrients between various particulate and dissolved components (i.e. not all phosphorus will be associated with phytoplankton), and by the potential for nitrogen deficiencies during actual growth tests (H. Hamilton, pers. communication).

DISCUSSION

The data gathered during the first year of study indicate that these waterbodies are still in their natural condition, with the watersheds of most lakes being essentially undeveloped. This pre-impact condition is generally characterized by moderate fertilities, moderate phytoplankton standing crops, and a natural summer oxygen depletion in the hypolimnia of most stratified basins. According to the chlorophyll and phosphorus ranking criteria of Likens (in Leith and Whittaker, 1975) the lakes encompass a trophic range from oligo-mesotrophic (Cold Lake) → meso-eutrophic (Ethel, May, Marie, Hilda, Moore, Bourque, Primrose South) → eutrophic (Tucker, Primrose North).

The higher fertilities of the eutrophic lakes are probably related to their shallowness, which in turn enhances the potential significance of internal loadings in their annual nutrient supply. The remainder, however, are probably more influenced by external loadings, and as such would be quite vulnerable to watershed land use changes.

At present these lakes have a high amenity value, and if protected can serve as a source of recreation and pleasure for many generations to come. However, the existing phosphorus loadings can easily be altered by uncontrolled runoff and disruption to natural surface water drainages associated with large scale land clearing operations. Evidence now exists to suggest that the long-term export of phosphorus from a watershed may increase three to five-fold per unit land area as a result of clearing (Trew et al., 1981).

Consideration of these facts suggests that watershed disturbances will affect different lakes to different degrees, and that each lake should be managed on an individual basis. Furthermore, stringent drainage controls should be implemented on all future industrial plant sites and roadways/utility corridors, in order to minimize sediment transport into these aquatic systems.

Future water quality monitoring in this region will include a survey of the alkalinity/buffering capacities and salinities of the standing water resource throughout the Beaver River Watershed. At present, we believe that most waterbodies in this area are well buffered and not highly sensitive to acid precipitation. However, a survey will be conducted in the spring of 1982 to define any sensitive areas, and also to clarify the extent of the intrusion of the saline lake zone into this watershed.

The underlying assumption of our approach to eutrophication control is that phosphorus is the key factor which limits the amounts of phytoplankton biomass that will develop in most lakes. Data presented elsewhere (Schindler, 1976; Trew et al., 1981) suggest that even in lakes which are apparently nitrogen-deficient enough nitrogen can be derived from other sources (i.e. N-fixation) to permit phosphorus - proportional growth to occur.

However, certain data from the Cold Lake regional lakes indicate relationships between nitrogen supplies and primary production. In order to clarify the role of nitrogen further, biostimulation experiments and ^{32}P-uptake experiments will be conducted throughout 1981.

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APPENDIX N

IMPACTS OF LAKE LEVEL FLUCTUATIONS ON TROPHIC STATUS

A Report Prepared By
D.O. TREW
ALBERTA ENVIRONMENT
POLLUTION CONTROL DIVISION
WATER QUALITY CONTROL BRANCH

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Pollution Control Division

Oxbridge Place, 9820 - 106 Street, Edmonton, Alberta, Canada T5K 2J6 403/427-6270 Environment Emergency 1-800-222-6514

Sept. 8, 1983

Mr. Ron Hicks
Athabasca River Basin Planner
Planning Division
Alberta Environment
9820 - 106 Street
Edmonton, Alberta
T5K 2J6

Dear Mr. Hicks:

We are pleased to submit the following report entitled "An Approach For Assessing the Impact of Lake Level Fluctuations on Lake Trophic Status".

The report outlines the probable impacts of industrial/municipal water withdrawals on the water quality of thirteen lakes in the Beaver River-Cold Lake drainage systems. A qualitative sensitivity index based upon morphometric, hydrologic, and edaphic characteristics was developed to rank the lakes according to their vulnerabilities to water withdrawals.

Yours truly,

A/DIRECTOR

R. N. Briggs, M. Sc., P. Eng.
for Director
Pollution Control Division

DT/sl

BACKGROUND

In 1980 the Planning Division of Alberta Environment initiated the Beaver River/Cold Lake Regional Water Resources Study. One of the objectives of the study was to assess the existing water supply/water demand situation in the basin, and also to assess various regional development scenarios with respect to the potential impact on water resources.

As part of this study, thirteen lakes in proximity to proposed or existing oil sands pilot plants were selected for investigation as potential industrial water sources. These lakes were: Caribou, Reita, Tucker, Ethel, Manatokan, Moose, Muriel, Angling, Burnt, Wolf, May, Marie, and Cold. The Water Quality Control Branch of Pollution Control Division was asked to assess the impact of artifically raising or lowering lake levels by up to one meter as a result of water withdrawals.

INTRODUCTION

The prediction of water quality changes resulting from artificial lake level fluctuations is a technical problem which has not received much attention in the limnological literature, except in the area of reservoir management; even there the information is limited. The "trophic upsurge" phenomenon frequently observed in new reservoirs has been reviewed by Ostrofsky (1978). Some theoretical considerations of nutrient leaching from inundated terrestrial detritus, and the effects of drying/freezing of exposed lake sediments were discussed in his paper. Although other chemical changes in new reservoirs have recently been documented, most notably the bio-accumulation of mercury in fish (Noton 1982), our comments will pertain principally to eutrophication and how this process might be affected by hydrologic manipulations.

This approach has been taken because many of these thirteen study lakes have a high amenity value, and recreation is probably the most important use of these resources at present. Furthermore, recreational pressures will increase as the provincial and regional populations continue to grow, placing more demands on the higher quality lakes. Accelerated eutrophication of these lakes either through impacts to the phosphorus loading or hydrological regimes could affect those characteristics of the lakes which are and will continue to be the most highly regarded for the recreational experience. These include: the aesthetic appeal of clean, intact shorelines; absence of exposed, decomposing vegetation and sediments; high clarity of the water; ease of lake access and boat launching; good fish production, etc. Furthermore, an eutrophied system has a negative impact for industrial/municipal use by clogging filters of water intakes and causing general taste and odor problems (e.g. Moose Lake).

Our major concern from a eutrophication viewpoint is the increase in nutrient supply through the leaching of nutrient material from either freshly inundated shoreline or from the exposure of organic sediments in the littoral zone during drawdown. The impact of the former will depend on the amount of soil, leafy vegetation, wood, etc. which is transported into the water column by flooding. This will act as a net nutrient input and its overall significance will depend upon the areal extent of flooding but also upon the edaphic characteristics of the backshore areas. Similar comments apply to drawdown: the enhanced release of nutrients following dessication/freezing will serve as an additional source to the lake, the severity of which will depend both upon the area and composition of the sediments exposed (Cooke 1980). The seasonal timing and duration of these manipulations may be a critical consideration. Finally, the extent of mobilization of mercury and other materials from recently flooded soils will be in part proportional to the surface area involved, and in that sense the severity of those impacts can be assessed using a similar approach to that described below for eutrophication.

A second important factor is the direct impact on the littoral zone, which is an important biological and chemical processing component of the lake ecosystem. In many cases, the littoral zone may the the major determinant of the annual primary production in a lake (Wetzel et al. 1972). It is the site of the highest biomass and diversity of benthic invertebrates, and because of these and other factors it is usually the major fish habitat (Hanson and Leggett 1982). Furthermore, because of elevated water and substrate temperature, macrophyte growth, and because all surface nutrient inputs contact the littoral zone first upon entering the lake, it is a site of considerable nutrient cycling; evidence now suggests that littoral zone macrophytes may effectively act as phosphorus "pumps" to the pelagial zone (Carignan and Kalff 1981; Carpenter 1980).

The biological processes of the littoral zone are controlled by the physical/chemical environment within it, which in turn is partly determined by the natural fluctuation in lake levels. By artifically increasing the range of this natural regime through short-term vertical fluctuations on the order of one meter, we anticipate direct perturbations to the physical/chemical characteristics of the littoral zone and therefore to its biota.

A critical feature of the aquatic habitat is oxygen depletion during periods of thermal stratification. This process has direct implications for fish production and indirect implications for nutrient cycling. The rate of oxygen depletion will be dependent upon the supply of biochemically/chemically oxidizable substrate, and physical conditions in the water column and at the sediment/water interface. An increase in this supply in certain lakes may elevate the oxygen demand to the point where winterkill situations could develop. Since the protection of fish habitat is an important consideration for both recreational and commercial interests, a systematic analysis of winterkill risk as affected by hydrologic manipulations will also be undertaken.

METHODS

It is proposed that these lakes be ranked according to their sensitivities to hydrologic manipulations, with the recommendation that only the least sensitive systems be considered for use. The following analysis will be based upon the absolute physical/chemical characteristics of the study lakes, with no interpretations regarding their existing recreational, aesthetic, or fisheries value included.

We have derived ranking criteria based upon the two principal considerations in eutrophication modelling: the effects of nutrient supply and the effects of flushing. This approach will also accommodate our concerns with respect to the relative ecological vulnerability of the littoral zones in question. While we are unable to present quantitative predictions regarding nutrient change, it is obvious from the preceding discussion that some lakes will be more sensitive than others, depending on certain attributes of their littoral zones and backshore areas. We propose that lakes with larger littoral zones in proportion to total surface area be ranked as highly sensitive on the basis that larger areas of sediments will be exposed, and that larger backshore areas will probably be flooded. The chemical and physical structure of the sediments will influence the results of exposure to the atmosphere and freezing. Fine silt or organic deposits will be more reactive than cobble/sand deposits; therefore the lakes with higher proportions or organic sediments will be ranked more sensitive.

Changes in nutrient availability in terms of mass supplied per unit time may be modified by the effect of flushing. Generally speaking, lakes with a higher rate of flushing, or shorter residence times will be affected less by loading changes (Dillon 1975).

The littoral zone (defined here as the shoreward region bounded by the maximum depth of colonization of macrophytes) has arbitrarily been defined at the 4.0 m depth contour. Naturally there will be exceptions, but this figure is based upon local experience in Alberta lakes (J. Crosby, Planning Division, pers. comm.). In eutrophic lakes where light penetration is limited by high standing crops of phytoplankton, the zone may be shallow (e.g. Tucker Lake) whereas in an unproductive waterbody such as Cold Lake the vertical light extinction is much lower and deeper colonization may be possible. In any event, there is also some evidence for hydrostatic pressure restrictions to colonization regardless of light penetration (Moss 1980). An expression presented by Spence (1976) for the relationship between the euphotic zone of phytoplankton (z_{eu}) and the colonization depth of macrophytes (z_c) for some Scottish lochs suggests that for euphotic depths of 8-10 m, which are common to many of the deeper lakes in this study area (Trew et al. 1981), a littoral zone depth of approximately 4 meters could be derived:

$$z_c = 0.37\ z_{eu} + 1.44$$

The relative sizes of the littoral zones were defined by comparing the surface areas between 0 m and 4.0 m with the total lake area. This data were interpolated between contour depths from the area-capacity data.

The proportions of organic sediments in the littoral zones were determined planimetrically from the shoreline maps prepared by C. McGregor, Alberta Energy and Natural Resources. The length of shoreline adjacent to aquatic or emergent zones dominated by organic/silt deposits was compared to the total shoreline length to derive a percentage.

All hydrology data were prepared by Hydrology Branch, Alberta Environment. The existing, long-term residence time (filling time) was calculated for each lake as was average annual water outflow.

Finally, an analysis of mean depths (z) can be used as a guide to determine which lakes are most sensitive in terms of winterkill potential. Barica and Mathias (1979) provide a technique whereby mean depth, the dissolved oxygen storage at freeze-up, and a dissolved oxygen depletion rate of 0.3 g m^{-2} day^{-1} can be utilized to predict the winterkill risk in productive prairie lakes. Preliminary information on winter oxygen depletion in local Alberta lakes (Babin 1982) indicates a similar depletion rate may be applicable. Therefore an approximation or winterkill risk can be found from a knowledge of mean depth alone, with assumptions regarding the level of saturation at freeze-up (100%). According to this approach, lakes with a mean depth less than 2.0 m are in a regular winterkill mode, lakes within a depth range of approximately 2.0 to 3.0 m occasionally winterkill, and lakes with a mean depth greater than 3.0 m have little risk. However, these are only approximations, and a conservative approach to their application would be wise until better local information is available.

RESULTS AND DISCUSSION

The extent and composition of the littoral zones of all lakes are summarized in Table 1. The percentage figures for each characteristic have been combined to produce an arbitrary numerical index indicative of the potential for increased nutrient supply and littoral zone damage.

TABLE 1 LITTORAL ZONE CHARACTERISTICS

	LITTORAL ZONE	% ORGANIC/SILT	TOTAL
CARIBOU	100	100	200
REITA	100	100	200
TUCKER	70	100	170
ETHEL	40	100	140
MANATOKAN	60	70	130
MOOSE	50	60	110
MURIEL	20	80	100
ANGLING	40	50	90
BURNT	50	30	80
WOLF	40	30	70
MAY	40	30	70
MARIE	20	10	30
COLD	10	20	30

According to this ranking system approximately three divisions of sensitivity can be seen: less than 70 (low), 70-140 (medium), and 140-200 (high). The large, deep lakes appear to have the least potential for increased nutrient supply or negative impact to the littoral zone, whereas the small, shallow lakes have the highest potential. This is because of the relatively large proportion of littoral zone to total lake area in the shallow systems, and also because of the fact that the organic sediments have not been transported to deeper, pelagial zones as is frequently the case in stratified lakes. The intermediate group, by coincidence, appear to be the lakes of highest recreational/sportfishing activity.

This sensitivity ranking also parallels the general trophic pattern (Water Quality Control Branch, unpublished data), indicating that the (shallow) productive lakes are relatively more sensitive. Shallow lakes with larger littoral zones tend to be more productive in part due to their efficiency in recycling nutrients from the sediments (Fee 1980).

The key hydrologic factors [annual outflow Q ($dam^3\ yr^{-1}$), T_W (residence time, yr), and z (mean depth, m)] are summarized in Table 2.

TABLE 2 HYDROLOGIC PARAMETERS

LAKE	ANNUAL OUTFLOW Q ($dam^3\ yr^{-1}$)	RESIDENCE TIME T_W (yr)	MEAN DEPTH z (m)
CARIBOU	133,713	0.1	0.9
COLD	561,341	22.9	49.5
MAY	10,880	1.3	5.6
ETHEL	17,400	1.6	6.6
MANATOKAN	16,275	0.7	2.7
BURNT	8,027	1.6	3.7
WOLF	51,828	4.1	8.7
TUCKER	11,935	1.3	2.9
REITA	0	4.4	2.5
ANGLING	4,985	1.7	5.5
MOOSE	24,558	4.6	5.3
MARIE	14,562	14.5	14.1
MURIEL	13,350	9.4	6.6

The range of water outputs and residence times encompass three orders of magnitude, and seem to produce somewhat different indices of sensitivity than were indicated by the analysis of littoral zones. For instance, Caribou Lake, which was ranked as highly sensitive in terms of potential nutrient increases, is seen to be the least sensitive according to flushing criteria. Similarly, Cold and Marie Lakes were the least sensitive according to their morphoedaphic characteristics but have the highest residence times - theoretically making them more vulnerable to increased phosphorus loads.

It is clear that the available eutrophication modelling methodology is inadequate for assessing this type of impact, since there is no means by which qualitative data on nutrient supply can be combined with quantitative data on hydrology to predict change. However, one point is clear: the compensating effects of flushing on nutrient loading, where they exist, will have little mitigating effect on damage to the biota of the littoral zone. In this regard, the littoral zone sensitivity ranking in Table 1 which theoretically accommodates these other concerns should be regarded as the most important criteria.

In terms of winterkill risk, it is obvious that several lakes would be jeapordized by a 1.0 m drawdown. Caribou appears to be in a "permanent" winterkill situation now, although the high flushing rate may modify its sensitivity. Manatokan, Tucker, and Reita are in the "occasional" winterkill category at present (2.0 - 3.0 m), but would probably winterkill permanently if drawndown by 1.0 m. Burnt Lake could drop into the "occasional" winterkill status. Angling, Moose, and May Lakes all have mean depths between 5.0 and 6.0 meters; allowing for some latitude in the prediction due to regional and individual lake variabilities, as well as short-term climatic trends, these lakes should be cautiously considered for use. Thus, the shallow lakes as a group are more sensitive to manipulation than the deep lakes according to their winterkill potential. Although there is a positive correlation between oxygen depletion rates and depth according to Barica and Mathias (1979), the slope of their regression is quite low indicating that while depth can increase significantly only small increases in the areal oxygen depletion rate occur. Shallow lakes have much less storage capacity at freeze-up (expressed on an areal basis) and therefore deplete sooner.

An analysis of annual water outflow offers a fourth criterion to assess the problem. One immediately observes that Caribou and Cold

Lakes have significantly higher discharges than any of the other study lakes. Therefore, the quantities of water that would have to be removed from these two lakes to effect a one meter drawdown are significantly larger than would be the case on the other lakes with smaller hydraulic loadings.

CONCLUSION

An examination of the littoral zone characteristics, flushing rates, winterkill risk, and outflow volumes of the 13 study lakes suggests that all lakes could be impacted by in-lake water withdrawals. Based upon the assumption that the existing trophic status is the preferred one, and that any change from that natural condition is undesirable, these impacts would be viewed as negative. However, the absence of quantitative estimates of nutrient loading changes permit only a qualitative, comparative assessment of the impacts.

Generally, the shallow productive lakes with large littoral zones will be more sensitive than the deeper lakes with small littoral zones. Although the effects of altered nutrient supply in the shallow lakes can be offset somewhat by their higher flushing rates, damage to the biota of the littoral zone cannot. The probability of winterkill in the shallow lakes will be increased by drawdown. The morphoedaphic characteristics of Cold Lake and its large outflow suggest that this lake would be the most suitable for water withdrawals.

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APPENDIX O

RIVER WATER QUALITY ASSESSMENT

A Report Prepared By

MACLAREN PLANSEARCH : LAVALIN

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20 September 1982

Mr. R. Hicks
Athabasca River Basin Planner
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Alberta Environment
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Edmonton, Alberta
T5K 2J6

Cold Lake - Beaver River Regional Water Resources Study
Water Quality Component

Dear Mr. Hicks:

We are pleased to submit our report on the effects of future developments upon the streams of the Beaver River Basin. The primary tool used in carrying out these predictions was the stream quality model, Qual-II.

We hope this report meets with your approval and look forward to further involvement in the planning of orderly development in the Beaver River Basin.

Yours very truly,

MACLAREN PLANSEARCH SERVICES LTD.

J. Penel, P. Eng.
Project Manager

JP/js

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SUMMARY

The stream quality model, Qual-II, was selected to describe the existing water quality and to predict the future changes in the streams within the prescribed study area of the Beaver River basin. This model was calibrated using data collected during a sampling survey conducted by Alberta Environment in August 1981. This sampling survey revealed that the streams of the basin were predominantly of very good water quality. Muriel Creek illustrated the effects of agricultural discharges along its course and municipal discharge to its headwaters.

A second data set, collected during September 1981 during lower flow period than the first survey, was used to verify the accuracy of the calibrated model. Variance between the predictions of the calibrated model and the data of the second survey were noted. These differences were not considered critical, but reflected the degree of confidence which may be placed in the future water quality predicted for the basin.

A total of four model runs were carried out to predict the effects of industrialization and the accompanying population increase upon the water quality of the basin. These were carried out using the low flow conditions generated from historical data, and an ultimate development scenario to the year 2000 A.D. provided by Alberta Environment. This development scenario included the installation of two large heavy oil plants and ten medium sized ones, plus a population increase of 144%.

The results of these runs indicated that current water quality criteria will be exceeded for the following parameters:

- un-ionized ammonia
- chloride
- sodium
- sulphate
- phenol

These criteria will be exceeded in Marie Creek, Muriel Creek and the Beaver River below Marie Creek. Discharges of wastes from heavy oil upgrading plants will be responsible for the excessive levels of chloride, sodium and sulphate above Marie Creek also. Phosphorus, un-ionized ammonia and phenol criteria will be exceeded due to discharges of treated municipal wastewater to Marie and Muriel Creeks. These excessive levels will be carried out into the lower Beaver River, where the criteria will also be exceeded.

Discharges of municipal wastes to Muriel Creek from Bonnyville and Ardmore-Fort Kent will result in critically low dissolved oxygen levels in the stream. Municipal waste discharges to Marie Creek may, in conjunction with the diurnal photosynthesis cycle, result in critically low dissolved oxygen values.

1.0 INTRODUCTION

Extensive industrial development is anticipated in the Beaver River basin to exploit the oil sands formations which underly the area. This development will take the form of heavy oil upgrading plants which produce synthetic crude oil for transport to refineries as feedstock. This industrial development will lead to a large influx of population to the basin. The anticipated doubling of the population and the industrialization will place pressure upon the water resources of the basin to supply water and assimilate wastes.

The maintenance of high standards of water quality is important in both the provincial and inter-provincial context. Accordingly, both Alberta Environment and the Prairie Provinces Water Board have promulgated water quality criteria which apply to the Beaver River system. Proposed industrial and municipal developments must be carried out so that these minimum quality levels are maintained.

Some impact upon local receiving waters is inevitable when large changes in basin land use, such as those proposed for the Beaver River basin, occur. With careful advance planning the adverse effects of development upon waterways can be reduced to acceptable levels. An important tool in this planning process is the realistic estimation of future waste discharges and their effects upon the receiving water. Accordingly, the commissioning of a study to estimate the water quality of the basin after industrialization is timely and appropriate.

The two primary objectives of the study were:

1) To describe the existing water quality of the streams in the Beaver River basin, and the mechanisms affecting them, and

2) To predict the change in basin water quality which would result from the proposed industrialization of the basin.

2.0 BASIN DESCRIPTION

The Beaver River Basin is located in east-central Alberta and forms the headwaters of the Churchill River system which flows eastward to Hudson Bay. The basin area in Alberta is approximately 15300 square kilometres and it extends about 160 km from east to west and 150 km from north to south.

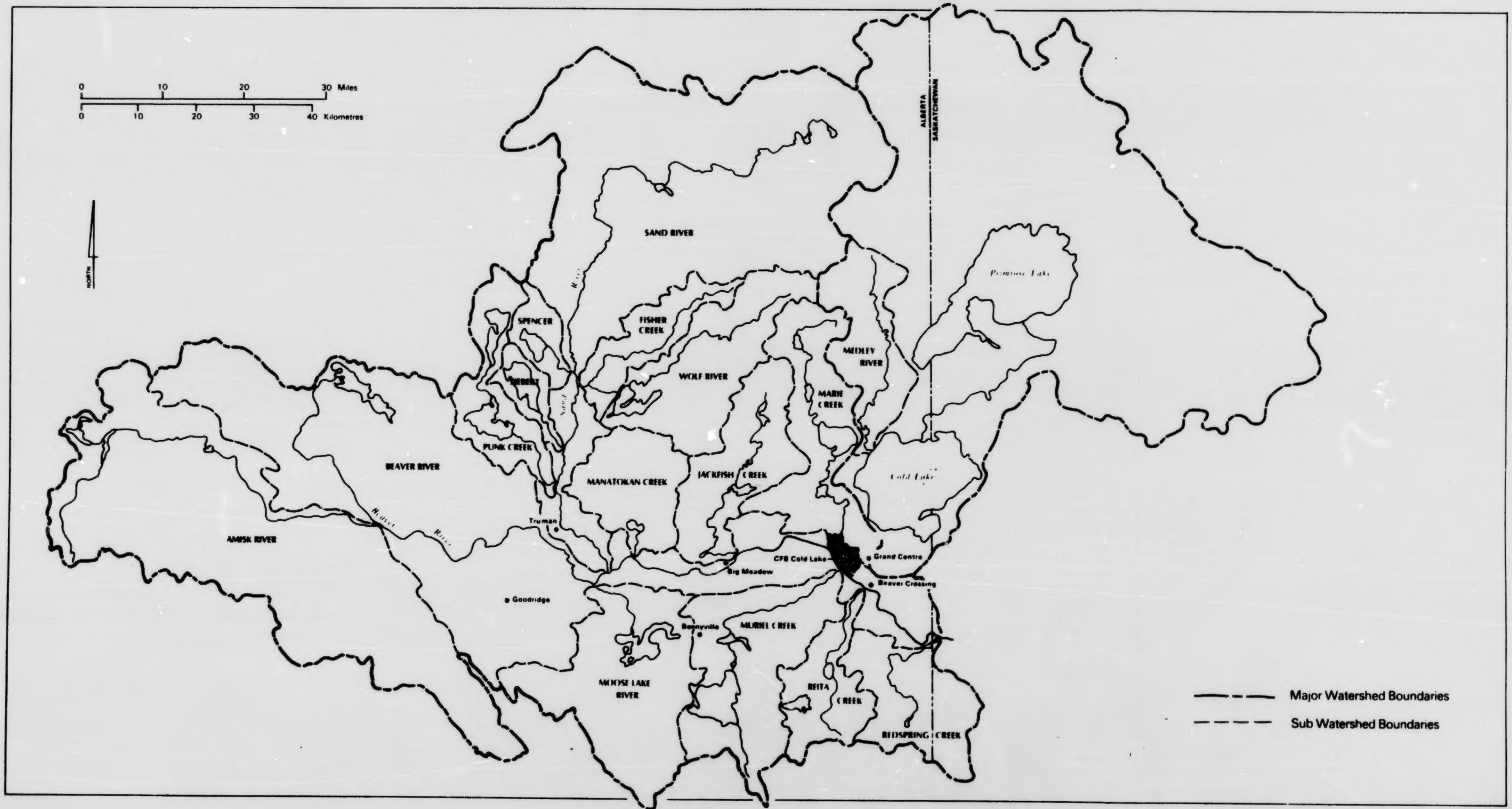
The study area for the current investigation extended from the Goodridge gauge on the Beaver River, along its length to the Alberta-Saskatchewan border. The Sand River was included from the outlet of Caribou Lake to its confluence with the Beaver River. Other tributaries included in the study area were, Mooselake River, Manatokan Creek, Jackfish Creek, Marie Creek, Muriel Creek and Reita Creek.

The Beaver River Basin and major sub-basins are illustrated in Figure 2-1. The major tributary is the Sand River, which flows into the Beaver River from the north. It is normally larger than the Beaver above the confluence and contributes up to five times the annual Beaver River streamflow in the upper reaches. The Sand River originates in the Primrose Air Weapons Range and flows southward through the gently rolling Mostoos Hills. Wolf, Spencer, Touchwood, Siebert and Pinehurst Lakes drain into the Sand River.

The Medley River, Marie Creek, Jackfish Creek and Manatokan Creek drain the south slopes of the Mostoos Hills into the Beaver River from the north. Mooselake River, Muriel Creek, Reita Creek and Redspring Creek flow into the Beaver River from the south where a low divide separates the Beaver River

Figure 2-1

Beaver River Basin Study Area


0 10 20 30 Miles
0 10 20 30 40 Kilometres
NORTH
ALBERTA
SASKATCHEWAN
SAND RIVER
Sand River
SPENCER
FISHER CREEK
HEBERT
WOLF RIVER
MEDLEY RIVER
MARIE CREEK
Primrose Lake
PUNK CREEK
BEAVER RIVER
MANATOKAN CREEK
JACKFISH CREEK
Cold Lake
Beaver River
Truman
AMISK RIVER
CFB Cold Lake
Grand Centre
Big Meadow
Beaver Crossing
Goodridge
Bonnyville
MURIEL CREEK
MOOSE LAKE RIVER
REITA CREEK
REDSPRING CREEK
Major Watershed Boundaries
Sub Watershed Boundaries

32782-0

and North Saskatchewan basins. The outflow from Cold Lake flows eastward into Saskatchewan as the Waterhen River and eventually joins the Beaver River upstream of Lac Ile a la Crosse.

Upstream of the Sand River, the Beaver River Basin is characterized by rolling hills, numerous small lakes and poorly drained, marshy areas. The rolling topographical features of the basin are largely remnant from the last glaciation. Ground moraine is interspersed with small pockets of glaciofluvial and glaciolacustrine deposits near existing lakes. The Sand and Beaver River Valleys are defined by old glacial meltwater channels.

The Beaver River Basin marks the northern frontier of agricultural land use in the eastern portion of Alberta. New land is currently being cleared as the land is patented. The general land use pattern is summarized in Table 2.1.

The Mooselake River watershed has the most intensive agricultural use. Approximately 50 percent of the land is used for grain, pasture and hay crops. Approximately 40 percent of the Muriel Creek watershed is used for agriculture. Along the Beaver River between the Sand River and Muriel Creek, about 30 percent of the land is cleared for agricultural use, while above the Sand River about 15% of the land is used for agriculture. In the rest of the basin, less than 5 percent of the land is used for agriculture.

TABLE 2.1

GENERALIZED LANDUSE PATTERN FOR THE BEAVER RIVER BASIN

	Percentage of total area	
Sub-basin	Agriculture	Forest/Wetlands
Sand River	1	99
Marie Creek	5	95
Muriel Creek	40	60
Mooselake River	50	50
Beaver River		
- above Sand River	15	85
- between Sand River and Muriel Creek	30	70
- below Muriel Creek	5	95

Source: MacLaren Plansearch 1981. Withdrawal Water Demands, Cold Lake Beaver River Basin.

The remainder of the basin is uncleared. The cover consists predominantly of aspen and balsam poplar. Jack pine is found on well-drained sandy sites while black spruce and tamarack are found in wet areas. Drainage is poor in the northern portion of the study area, where muskeg wetlands predominate.

The 1980 population of the Beaver River Basin was approximately 26 500 (Table 2.2). An urban population of almost 10 000 inhabits four urban centres: Bonnyville, Cold Lake, Grand Centre and Glendon. CFB Cold Lake has a population of over 5000. Approximately 2500 people live on four Indian Reserves in the Basin. The remainder of the population lives in smaller towns or rural areas.

Industrial activity in the Beaver River Basin is focused on experimental or pilot plants related to oil sands extraction. There are thirteen companies currently operating experimental projects or which may in future operate an experimental project. The sub-basin location and operating companies are summarized in Table 2.3.

TABLE 2.2

BEAVER RIVER BASIN POPULATION 1980

Bonnyville	4 256	
Cold Lake	1 648	
Grand Centre	3 107	
Glendon	454	
Total Urban	9 465	9 465
C.F.B. Cold Lake	5 500	
Ardmore	236	
Fort Kent	140	
Other Rural	4 606	
Total Municipal District at Bonnyville	10 482	10 482
Total County Athabasca	276	276
Mallaig	200	
Rural	1 290	
Total County St. Paul	1 490	1 490
Total Country Smoky Lake	247	247
Total I.D.	2 055	2 055
Beaver Lake I.R.	263	
Cold Lake I.R.	1 019	
Kehewin I.R.	695	
Whitefish Lake I.R.	569	
Total Indian Reserves	2 546	2 546
TOTAL		26 561

Source: MacLaren Plansearch, 1981. Withdrawal Water Demands, Cold Lake Beaver River Basin.

TABLE 2.3

CURRENT AND FUTURE OIL SANDS PROJECTS FOR THE BEAVER RIVER BASIN

Sub-basin	Operating Company	Experimental Project Current	Experimental Project Possible Future
Sand River	Amoco/Kissinger	x	
	Canada - Cities Service	x	
	British Petroleum	x	
	Dome		x
Manatokan Creek	Mobil		x
Marie Creek	Esso Resources	x	
Muriel Creek	Worldwide/	x	
	Petro Canada	x	
Cold Lake	Norcen	x	
	Gulf Canada Resources		x
Beaver River	Chevron	x	
	Union Texas	x	
	Great Northern	x	

Source: MacLaren Plansearch, 1981. Withdrawal Water Demands, Cold Lake - Beaver River Basin.

3.0 WATER QUALITY MODEL DESCRIPTION

3.1 General

QUAL-II is a comprehensive stream water quality model which can simulate up to 13 water quality constituents in any combination desired by the user. Constituents which can be simulated are:

1) Dissolved Oxygen
2) Biochemical Oxygen Demand (BOD)
3) Temperature
4) Algae as Chlorophyll a
5) Ammonia as N
6) Nitrite as N
7) Nitrate as N
8) Dissolved Ortho-phosphate as P
9) Coliforms
10) Arbitrary Non-conservative Constituent
11) Three Conservative Constituents

The model is designed to be applied to dendritic well mixed streams and assumes that the major transport mechanisms, advection and dispersion, are significant only along the main direction of flow. It allows for multiple waste discharges, withdrawals, tributary flows, and incremental inflow and has the capability to compute required dilution flows for flow augmentation to meet any prespecified dissolved oxygen level.

QUAL-II can be operated as a steady-state model or a dynamic model. In the steady state mode QUAL-II is limited to the simulation of periods during which the stream flows vary no more than 10 to 15 percent. Input waste loads must also be held constant. Dynamic operation makes it possible to study water quality (primarily dissolved oxygen and temperature) as it is affected by diurnal variations in meteorological data. A detailed description of the model is provided in the documentation provided by the U.S. Environmental Protection Agency (1981a, 1981b).

The model permits branching, one-dimensional stream systems to be simulated. The first step involved in approximating the stream is to subdivide the system into reaches, on the basis of hydraulic characteristics. Each reach is then divided into computational elements of equal length throughout the system. Simulation computations are carried out in each computational element, assuming a completely mixed regime and the products transported to the next element by advection.

The number of reaches and computational elements are limited as follows:

Reaches: a maximum of 75
Computational elements: no more than 20 per reach or 500 in total
Headwater elements: a maximum of 15
Junction elements: a maximum of 15
Input and withdrawal elements: a maximum of 90

3.2 Hydraulic Simulation

3.2.1 Stream Discharge

Discharge data at each reach can be input in the following manner:

- headwater inflow at the starting point of a river or tributary.
- incremental inflows to simulate natural groundwater and uniformly distributed runoff. Those inflows are assumed to be uniformly distributed over each reach.
- point source inflow or withdrawals.
- flow from tributaries

QUAL-II is not a hydraulic model, flow cannot be simulated and the discharge has to be re-adjusted manually for the various flow conditions to be studied in each case. A large flexibility in discharge input is possible, for example, a river tributary can be treated as a source input or a tributary input depending on the degree of data available and the specific reaches to be modelled. Final calibration adjustment of flow is done using incremental inflows.

3.2.2 Velocity and Water Depth

Two options are available to calibrate the velocity and water depth associated with discharge at each reach of the system:

1) velocity and depth are input as power functions of discharges. This option is normally used when river rating curves and cross-sections are known and can be written as a power function.

2) each reach is represented as a trapezoidal channel. The program compares the velocity and depth using Manning's Equation and the Newton-Raphson iteration method (Roesner et al., 1981b).

3.3 CONSTITUENTS SIMULATION

In order to quantify water quality associated with each new input of water, each water quality constituent of interest must have an associated value for each non-zero discharge data input, namely at headwater flow, incremental inflow, and point source inflow.

In the case of conservative constituents, the model simply calculates the concentrations associated with the various flow volumes encountered along the river. Since there is no decay, calibration consists of adjusting input loads from headwaters, point sources and incremental inflows.

In the case of non-conservative constituents, a decay function is added to the mixing process of the conservative constituents.

Calibration of these constituents is first done in a similar manner to conservative constituents by adjusting input loads, secondly, decay rates can be varied.

Some of the constituents like BOD and coliforms are independent of the other constituent concentrations. Other constituents like algae, ammonia, nitrite, nitrate and dissolved ortho-phosphate are highly interrelated and are usually calibrated last.

3.4 Model Application

3.4.1 Beaver River Network

The Beaver River watershed consists of the Beaver River and numerous tributaries. For modelling purposes, the system was subdivided as follows:

• Beaver River	16 reaches
• Muriel Creek	5 reaches
• Marie Creek	3 reaches
• Jackfish Creek	4 reaches
• Manatokan Creek	1 reach
• Sand River	9 reaches
• Mooselake River	1 reach

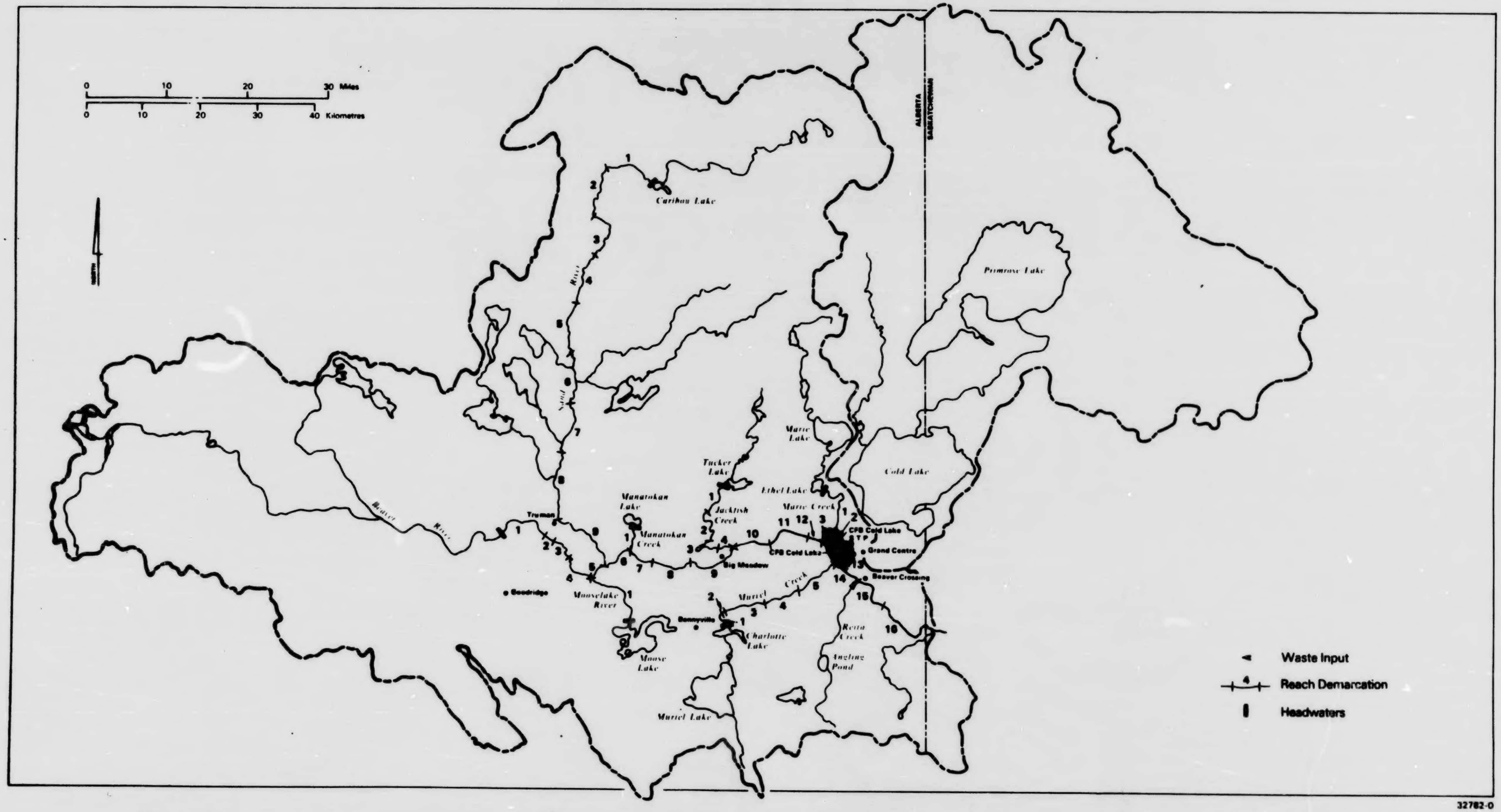
Since it will receive no large new waste discharges, for modelling purposes, Reita Creek has been treated as point source input. Figure 3-1 shows the locations of the reaches and waste inputs.

3.4.2 Constituents

The following is a brief description of each of the constituents that have been modelled in the Beaver River study using QUAL-II.

Figure 3-1

Reach Identification and Waste Inputs


0 10 20 30 Miles
0 10 20 30 40 Kilometres
NORTH
ALBERTA
SASKATCHEWAN
Caribou Lake
Sand River
Primrose Lake
Marie Lake
Tucker Lake
Ethel Lake
Marie Creek
Cold Lake
Manatokan Lake
Manatokan Creek
Jackfish Creek
Truman
Beaver River
CFB Cold Lake STP
CFB Cold Lake
Grand Centre
Big Meadow
Beaver Crossing
Goodridge
Mooselake River
Bonnyville
Charlotte Lake
Muriel Creek
Reita Creek
Angling Pond
Moose Lake
Muriel Lake
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
Waste Input
Reach Demarcation
Headwaters

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Dissolved Oxygen

Minimum dissolved oxygen concentrations of 3 to 5 mg/L are necessary to sustain a varied ecosystem. Concentrations of dissolved oxygen are reduced on a long term basis by organic oxygen demanding substances in the water column and in bottom sediments. These substances are input to the stream through runoff and organic waste discharges. The only source of oxygen in the stream is diffusion through the surface at a rate (K_2) determined by the degree of turbulence in the stream and the saturation concentration.

Biochemical Oxygen Demand (BOD)

BOD is the result of the instream decay of carbonaceous organic materials by micro-organisms. This decay exerts a demand for oxygen upon the stream which at times of high input can reduce the dissolved oxygen concentration to critical levels. Oxygen demanding substances originate in agricultural runoff, industrial and municipal waste discharges and natural runoff. Since there are no large point source discharges on the Beaver River the main sources of BOD will be agricultural and natural runoff.

Ammonia

Ammonia is a product of the bacterial decomposition of organic nitrogen forms in nature and is present in both industrial and municipal discharges. In water solution ammonia exists in the un-ionized (NH_3) form and as ammonium ion (NH_4^+) in a proportion governed by pH and temperature. Un-ionized ammonia is toxic to fish at concentrations as low as 0.02 mg/L (MacLaren, 1978). Ammonia also serves as a nutrient for certain bacterial species. Utilization by these species exerts an oxygen demand.

Nitrite (NO_2^-) and Nitrate (NO_3^-)

These are products of the bacterial oxidation of ammonia (nitrification). Nitrite is a short lived intermediate form which rapidly oxidized to nitrate. Nitrate is not significantly toxic to fish but nitrite does exhibit toxicity at concentrations as low as 0.2 mg/L. Both forms are important nutrients for aquatic plant growth. Nitrate levels of greater than 10 mg/L-N in water supplies have been associated with infantile methemoglobinemia.

Dissolved Ortho-Phosphate ($PO_4^=$)

Dissolved ortho-phosphate is an inorganic form of phosphorous found in water. It originates from mineral weathering and the hydrolization of unstable condensed phosphates. Condensed phosphates originate in both anthropogenic and natural discharges. Soluble ortho-phosphates are readily assimilated by plants and other aquatic organisms and in conjunction with other phosphate forms sometimes represent the limiting factor to the growth of these organisms. Concentrations of 0.02 mg/L may contribute to rapid growth in waters.

Total Coliform

Coliform bacteria originate from agricultural runoff and discharges of human waste materials from communities. They decay in natural waterways in an approximately first order fashion, at rates varying from 0.01 day^{-1} to 10 day^{-1} (Tetra Tech. 1978). The magnitude of these rates depends upon the physical and chemical conditions existing in the receiving waters. In this study, total coliform bacteria was simulated.

Phenol

Phenolic substances are of concern because of the potential for releases from the proposed heavy oil developments in the Beaver River basin. Phenols add taste to drinking waters at levels in excess of 0.005 mg/L (Health and Welfare Canada, 1978) and are limited to this level at the Alberta-Saskatchewan boundary.

Sulphate ($SO_4^=$) Chloride (Cl^-) and Sodium (Na)

These substances were modelled as conservative species. Sulphate is not conservative in anoxic conditions but since the Beaver system is highly aerobic it may be considered conservative. Each of these were assumed to originate in the runoff from agricultural landuse in the basin with a direct relationship between the degree of use and the amount of material in the runoff. Point sources of human waste discharges and groundwater inputs would also contribute these species to the receiving water. Each contributes to the overall dissolved solids balance in the water. Excessive quantities of sulphate and chloride make water unsuitable for drinking and irrigation.

Algae

Although not included in this study, QUAL II has the capability of simulating the behaviour of instream algae. By relating measured chlorophyll A values to algal growth potential, the effect of algae upon instream nutrient concentrations and diurnal dissolved oxygen variations are simulated. Since chlorophyll A is a relatively difficult measurement, it is important that very good techniques be used in its determination.

4.0 DATA ACQUISITION

4.1 Collection of Background Data

At the outset of the study, background data were collected in preparation for the field sampling program. The data was gathered from three sources:

- previous investigations of the Beaver River
- a reconnaissance field trip
- a comprehensive sampling program

The Alberta Research Council (ARC) has conducted investigations of the Beaver River in the fields of hydrology, water quality and sediment transport. Few of these have been published, however, ARC provided access to the file data for this study. A number of these investigations provided useful background data:

- 1978 dispersion study of the Beaver River, downstream from the Ardmore Crossing

- 1974 Beaver River cross-sections, surveyed by the Research Council

- calculations of slope and Mannings "n" in the vicinity of Cold Lake Reserve

- thalweg soundings in the vicinity of Beaver Crossing

- rating curves and surveyed cross-sections at Beaver Crossing

The active Water Survey of Canada stream gauging stations are listed in Table 4.1. The Water Survey provided rating curves and cross-sections for their gauging stations in the study area, and Alberta Transportation provided surveyed cross-sections at the major bridges crossing the Beaver River and tributaries.

4.2 Calculation of Travel Time and River Slope

Travel times for three selected flow levels of the Beaver River was calculated as follows:

1) The relationship between monthly average discharge of the Beaver River at Cold Lake Reserve (6AD006) and near Goodridge (6AA001) was determined graphically. The best fit line had the following equation:

$$Q_{\text{Cold Lake}} = 5.5\, Q_{\text{Goodridge}}^{0.74} \qquad (1)$$

2) Low, medium and high flows at Cold Lake Reserve were selected and equation (1) was used to calculate the corresponding flows for Goodridge:

	Cold Lake Reserve m^3/s	Goodridge m^3/s
Low	10.0	2.2
Medium	30.0	9.9
High	100.0	50.0

These values were plotted on a discharge-drainage area curve to allow the estimation of flow for any point between the two stations.

TABLE 4.1

STREAMFLOW GAUGING STATIONS IN THE BEAVER RIVER BASIN

Name	Station Number	Basin Area (km^2)
Beaver River at Cold Lake Reserve	6AD006	14 500
Sand River Near Mouth	6AB001	4 920
Beaver River Near Goodridge	6AA001	4 710
Amisk River Near Highway 36	6AA002	2 500
Moose Lake River Near Franchere	Alberta Environment	985
Wolf River at Outlet of Wolf Lake	6AB002	725
Manatokan Creek	Alberta Environment	439
Pink Creek Near Mouth	Alberta Environment	401
Reita Creek	Alberta Environment	298

3) The Hydrology Branch has developed an equation to describe the relationship between velocity and discharge of the Beaver River:

$$V = 0.18\ Q^{0.33}$$

4) Equation (2) was combined with the discharge-drainage area curve to estimate the velocity of various points along the mainstem. The calculated velocity in m/s, was then used to estimate travel time.

Velocity calculations were required to compare QUAL-II results with measured data during calibration. Travel time was used to schedule sampling during the sampling programme.

The water surface profile and stream bed profile were prepared using the available surveyed profiles and cross sections, which were tied to geodetic survey points, prepared by Alberta Transportation and the Water Survey of Canada. Data gaps were filled using 1:50 000 scale National Topographic Survey maps, with 25 foot contours.

4.3 Reconnaissance Field Trip

A reconnaissance field trip was made to the Beaver River Basin in June 1982 in order to familiarize the team members with the study area and with the morphology of the Beaver River and its tributaries.

Water Samples were taken for chemical and bacteriological analysis at 12 locations. Dissolved oxygen and temperature were recorded. Approximate cross-sections were measured at most locations and velocity was measured at the cross-sections. The sample locations and results are shown in Appendix A.

Photographs were taken upstream and downstream of each sample location and were used to estimate Manning's n for each reach.

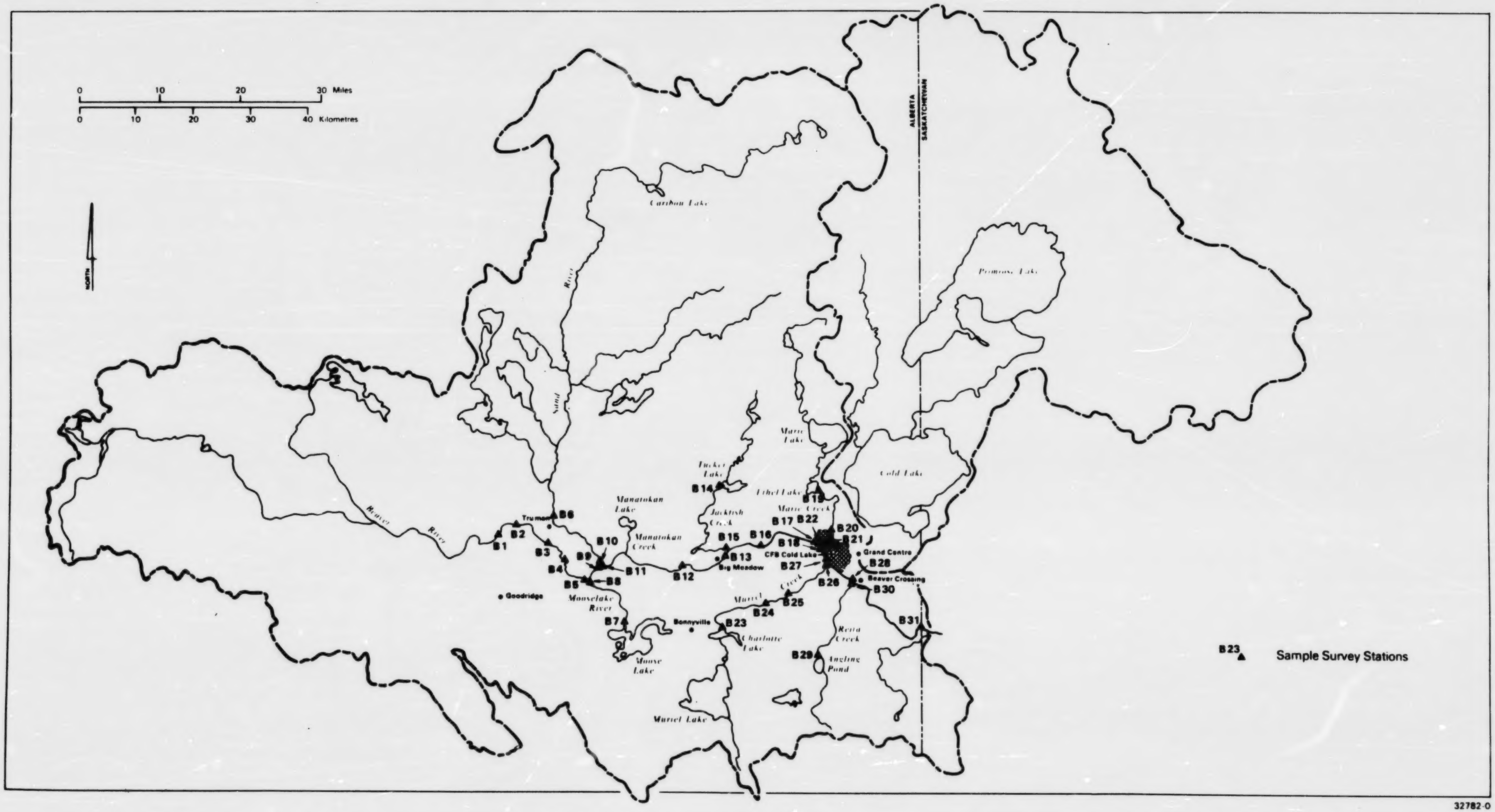
4.4 Sampling Program

Two intensive sampling programs were carried out during low flow periods in August and September. The sample program was developed by MacLaren Plansearch staff and the field work and analytical analyses were performed by Alberta Environment. Water samples were collected along the mainstream and tributaries of the Beaver River; as indicated in Figure 4-1. The sampling schedule was prepared to correspond with the estimated time of travel. Sampling took place over two to three days.

At each locations samples were taken for chemical and bacteriological analyses. Dissolved oxygen and water temperature, were also measured. Laboratory analyses for the following parameters were performed: BOD, $SO_4^{=}$, phenol, Na, dissolved ortho-phosphate, total phosphorus as P, NH_3, NO_2^-, NO_3^-, Cl^-, chlorophyll A, total coliforms, fecal coliforms. The analytical results of the two sample sets are detailed in Appendix A.

Figure 4-1

Sampling Survey Stations


0 10 20 30 Miles
0 10 20 30 40 Kilometres
NORTH
ALBERTA
SASKATCHEWAN
Caribou Lake
River
Sand
Beaver River
Primrose Lake
Marie Lake
Tucker Lake
Cold Lake
Ethel Lake
Manatokan Lake
Manatokan Creek
Jackfish Creek
Marie Creek
Truman
Goodridge
Mooselake River
Moose Lake
Bonnyville
Charlotte Lake
Muriel Creek
Muriel Lake
Reita Creek
Angling Pond
CFB Cold Lake
Grand Centre
Big Meadow
Beaver Crossing
B1
B2
B3
B4
B5
B6
B7
B8
B9
B10
B11
B12
B13
B14
B15
B16
B17
B18
B19
B20
B21
B22
B23
B24
B25
B26
B27
B28
B29
B30
B31
B23 Sample Survey Stations

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4.5 Discussion of Sample Results

The two sample periods were scheduled to coincide with low streamflow during the summer months. The mean discharges of the Beaver River at Highway 28, during the time of sampling were:

August 5 to 7 - 13.6 m^3/s
September 14 to 16 - 3.7 m^3/s

In general the results of the second, lower flow survey show higher instream concentrations of the parameters than these observed in the first survey. Certain trends became apparent in both surveys and are summarized below.

- Sodium - values were higher in the tributaries which flow into the Beaver River from the south. Muriel Creek had the highest values during the August survey, probably due to its route through a highly developed basin. No flow was observed in Muriel Creek during the September survey.

- Chloride - values were high in Muriel Creek and Marie Creek. This is probably due to the level of development in their basins.

- Sulfate - values are high for all northward flowing streams including Mooselake, Muriel, and Reita and very low for the southward flowing streams including the Sand, Marie and Jackfish Creeks. This is indicative of the higher level of development south of the mainstem.

- Dissolved Oxygen and Biochemical Oxygen Demand - values are usually within acceptable limits for the Beaver River and its tributaries. In Marie and Muriel Creeks, however, diurnal photosynthesis appears to reduce the dissolved oxygen concentration to below 5 mg/L.

- Dissolved Ortho-phosphate - values were generally high throughout the watershed and highest in Marie Creek.

- Ammonia and inorganic nitrogen - values were within acceptable limits and were considered low.

- Temperature - values were within the 18°C to 24°C range during the sampling periods.

- Coliforms - values are generally within acceptable limits, with several localized variations where the river is used for stock watering.

In summary, the Beaver River and its tributaries comprise a watershed relatively unaffected by the human discharges associated with urban and industrial development. With a few minor exceptions, most parameters are within the acceptable limits of the Alberta Surface Water Quality Guidelines.

5.0 MODEL CALIBRATION

5.1 Significance of Model Calibration

The calibration process is intended to demonstrate that the selected model actually describes the conditions and processes existing in the stream. The results of model simulations are compared to the field collected data; to improve the fit of the simulated system to the observed data, decay coefficients and non-point source inputs are adjusted within reasonable values.

Normally, decay coefficients are estimated from the observed data where decreases in constituent concentrations are identified downstream of a point load input. By assuming an appropriate kinetic formulation for the reactions, the decay constant can be computed. The calculation of accurate instream decay coefficients in this study was confounded by the absence of large point discharges on the mainstem.

The result is that for model calibration in a pristine system such as the Beaver River, a large number of combinations of decay rate and incremental input can be used to fit the model to the observed data. Experience and literature values must be used to ensure that reasonable values are chosen but the true value will be in doubt.

The thrust of the model calibration exercise therefore must be altered to reflect this situation. The sensitivity of the instream constituents to changes in decay rates must be determined so that a response envelope is identified. This sensitivity analysis is carried out by calibrating the model with the apparent decay rate, then altering the rate, within acceptable literature ranges to identify the error which may be expected.

In this study biochemical oxygen demand (BOD), dissolved oxygen, and coliforms were subjected to sensitivity analysis. The conservative species, chlorides, sulphate and sodium were calibrated using incremental inflow only and no sensitivity analysis was performed. Nutrients either behaved in a conservative manner or were present in very low concentrations. Accordingly, no sensitivity analysis was carried out on nutrients.

5.2 Calibration Procedures

The procedure followed in calibrating QUAL-II to the Beaver River system was to first calibrate the flow, followed by the quality parameters in ascending order according to the number of variables. In practice this means that the conservative or quasi-conservative substances are handled first. These are not degraded in the stream and concentrations are therefore dependent only upon flow. These materials include chloride, sulphate and sodium. Materials which undergo instream decay, non-conservative substances such as BOD, coliforms, nutrients and phenol are then calibrated.

5.3 Hydraulic Considerations

The depth and velocity of a river is affected by its slope and channel shape. In turn the depth, velocity and surface area of the stream affect the reaction times affecting in-stream constituents and the rate of reaeration. It is therefore of paramount importance to accurately represent the hydraulic regime before attempting the calibration of the quality parameters.

QUAL-II provides two facilities for modelling the steady state flow regime of the subject stream. (For practical purposes the steady state condition can be defined as a change of less than 15% in flow over the duration of the sampling period.) The first involves the use of discharge coefficients and exponents to compute the velocity and depth in the following form:

$$v = a\,Q^{b}$$

$$d = \alpha\,Q^{\beta}$$

where v = stream velocity (L^2/T)
Q = stream flow (L^3/T)
d = stream depth (L)

α, β, a, b = coefficients

Alternatively, the depth and velocity can be determined using the geometric properties of the channel by Manning's equation:

$$Q(ft^3/s) = \frac{1.486}{n} A_x R_x^{2/3} S_e^{1/2}$$

Where: n = Manning roughness coefficient

A_x = cross-sectional area of flow (feet)

R_x = hydraulic radius (feet)

S_e = slope of the energy grade line (unitless)

For the study of the Beaver River the latter form was selected.

The flow in the river is made up of contributions from the headwaters, tributaries, incremental inflow and waste discharges. The Beaver River was designated the mainstem and the Sand and Mooselake Rivers, Manatokan, Jackfish, Marie, Muriel and Reita Creeks were termed tributaries.

Data was collected during the field surveys at several locations within the system. Three Water Survey of Canada gauging stations are located on the rivers of the Beaver River basin; on the Beaver River at Goodridge and Beaver Crossing and on the Sand River at Truman. Additional data was collected at three locations on the mainstem using cross-section and velocity measurements. The data collected from the first survey is shown in Table 5.1.

There appears to be a lack of continuity between the Water Survey of Canada data and the data measured during the survey.

TABLE 5.1

Hydraulic Data

Location	Date	$Q(m^3/s)$	V(m/s)**	Mean Depth (m)
Beaver at				
Goodridge	Aug 3	1.63	-	-
Km 143	4	1.51	-	-
	5	1.44	-	-
	6	1.34	-	-
	7	1.25	-	-
Sand at	Aug 3	10.2	-	-
Truman	4	9.43	-	-
Km 18	5	8.67	-	-
	6	8.17	-	-
	7	7.75	-	-
Beaver at*	4	1.26	0.18	0.55
Hwy. 881				
Km 135				
Beaver at*	4	9.32	0.51	0.70
Big Meadow				
Km 78				
Beaver below*	6	10.7	0.42	0.50
Marie Creek				
Km 39				
Beaver at	Aug 3	16.7	-	-
Cold Lake	4	15.5	-	-
Reserve	5	14.5	-	-
(Beaver Crossing)	6	13.6	-	-
	7	12.6	-	-

* Measured during sampling survey No. 1
** Velocity calculated from instream measurements.

$$V = \frac{Q}{A}$$

This disagreement is probably due to the inaccuracy inherent in a single, spot measurement relative to that of a well established and calibrated stream gauge. To make the best use of the data, the Water Survey of Canada gauge data was used to identify the magnitude of the flows and the total incremental inflow between gauges, taking travel time into account. The survey data was used to apportion the incremental inflow data along the mainstem. Where applicable, known tributary flows were subtracted from the incremental inflow values. The incremental inflow volumes calculated for the mainstem using this method are shown in Table 5.2.

For the Sand River no survey flow measurements were made. Accordingly, incremental inflow was apportioned using a value typical of the upper reaches of the Beaver River. An estimate of the headwaters flow was made by back-calculating from the gauge at Truman, to Caribou Lake. The headwaters flow was assumed to be 5.84 m^3/s.

Headwater and incremental inflow data were not available for the other tributaries. These values were estimated from runoff - area correlations developed from historical data. Where the tributary was short, and the contributing area downstream of the headwater lake was small compared to that contributing to the headwater lake, the incremental inflow to the stream was assumed to be zero (Table 5.3).

Channel geometry was determined in the field by taking cross-sections at several locations along the mainstem and on tributaries during 1981. In addition, data collected by the Alberta Research Council in 1973 was used for the lower reaches of the Beaver. Side slopes varied from very flat to steep, as shown in Table 5.4.

TABLE 5.2

BEAVER RIVER INCREMENTAL INFLOW

From Km	To km	Reach No.	Q_I (m^3/s)
143	127	1	0.377
127	123	2	0.068
123	116	3	0.159
116	106	4	0.220
106	102	6	0.00
102	94	16	0.00
94	87	18	0.060
87	76	19	0.244
76	61	20	0.331
61	53	25	0.130
53	43	26	0.224
43	40	27	0.068
40	35	30	0.011
35	28	36	0.060
28	18	37	0.224
18	0.0	38	0.402

TABLE 5.3

TRIBUTARY FLOWS

Tributary	Headwater (m^3/s)	Incremental Inflow ($m^3/s/km$)
Mooselake River	0.009	0.0
Manatokan Creek	0.10	0.0
Jackfish Creek	0.05	0.0003
Marie Creek	0.09	0.0007
Muriel Creek	0.04	0.0003

TABLE 5.4

MEASURED CHANNEL GEOMETRY

Location	Side Slope (Horiz./Vert.)	Bottom Width (m)	Side Slope (Horiz./Vert.)
Mooselake River	8:1	22	16:1
Manatokan Creek	1:1	2.2	5:1
Jackfish Creek	1.1:1	3.8	2:1
Marie Creek	8:1	10.2	3:1
Muriel Creek	3.5:1	3	0:1
Reita Creek	1.5:1	.5	1.5:1
Beaver River Below Marie Creek	1.7:1	21	132:1
Beaver River Upstream at Jackfish Creek	45:1	12:7	2:1
Beaver River Above Manatokan	11:2	41.9	8:3
Beaver River Above Mooselake River	19:7	6	8:1
Beaver River Bridge D/s Truman Bridge	10:3	9	3:1
Beaver River Hwy. 881	4:1	19.6	6:1
Beaver River Between Truman and Goodrige Bridges	11:4	21:3	17:1

Channel roughness was estimated by visual comparison of field photographs and streams of measured roughness in Chow (1959) (Table 5.5). The predicted Manning's 'n' values were modified slightly to calibrate velocity. Overall agreement with illustrated literature values remained intact.

The results of the flow calibration on the mainstem is shown in Figure 5-1. Velocities throughout the mainstem agreed well with the measured values, ensuring accurate travel time calculation (Table 5.6). The Beaver River was found to be a relatively slow moving stream above the Sand River. However, with the inflow from the Sand, velocity increased. Average depth showed less accurate agreement. This was probably due to errors input when bottom irregularities are smoothed when specifying channel geometry.

5.4 Nutrients

Except for Marie and Muriel Creeks, the Beaver River system exhibits very low levels of dissolved inorganic nitrogen and high levels of total and dissolved phosphorous (0.15 and 0.033 mg/l, respectively). Subsequent investigations have suggested that the phosphorus concentrations determined during the two river surveys may have been erroneous due to the presence of organically imparted colour in the samples. Accordingly, the results of the calibration and prediction of phosphorus in the Beaver River must be interpreted very carefully until further data are available.

A convienent method of describing biological production in aquatic systems with respect to nutrients and productivity in the past has been the limiting nutrient concept. This stems

TABLE 5.5

BEAVER RIVER SYSTEM ROUGHNESS COEFFICIENTS, MANNINGS n.

REACH	ROUGHNESS COEFFICIENTS
1	0.028
2	0.028
3	0.028
4	0.028
5	0.050
6	0.035
7	0.040
8	0.040
9	0.040
10	0.040
11	0.040
12	0.040
13	0.040
14	0.040
15	0.040
16	0.028
17	0.055
18	0.035
19	0.035
20	0.035
21	0.050
22	0.050
23	0.050
24	0.050
25	0.028
26	0.028
27	0.028
28	0.050
29	0.050
30	0.028
31	0.050
32	0.050
33	0.050
34	0.050
35	0.050
36	0.028
37	0.028
38	0.028

TABLE 5.6

VELOCITIES AND DEPTHS

Location	Average Velocity (m/s)		Average Depth (m)	
	Measured	Simulated	Measured	Simulated
Beaver River at Hwy. 881	0.18	0.19	0.30	0.46
Beaver River at Big Meadow	0.5	0.42	0.37	0.97
Beaver River at Marie Creek	0.38	0.44	0.41	0.65

from Liebigs "Law of the Minimum" which states approximately that, under steady state conditions the essential material for growth which most closely approaches the critical minimum needed, will limit growth even if other materials are in abundance. Since phosphorous and nitrogen are major nutrients, the relationship between the two has often been used to describe the limitation state of aquatic systems. Growing systems require many substances as nutrients, those required in smaller amounts than nitrogen and phosphorous may limit production despite the apparent ratio between the two. It remains useful, however, to observe the phosphorous and nitrogen regime of a water body to explain various phenomena. The typical total N:P ratio for a phosphorous limited system is equal to or greater than 5:1. A comparison of the total P and total kjeldahl nitrogen values for the Beaver River shows that upstream of the Sand River the ratio was between 3 and 5, reflecting a mild nitrogen limitation. The very low levels of ammonia and nitrate nitrogen tend to support this. The Sand River added proportionately less total phosphorus than total nitrogen to the Beaver River shifting the ratio into the phosphorus limitation range. However, the ratios of total inorganic nitrogen to total inorganic phosphate (TIN/TIP) remains less than 1, both in the Sand River and over the entire length of the Beaver River. The overall balance of available nutrient forms, therefore remained virtually unchanged, and the low value of TIN/DOP implies nitrogen limitation. Under these conditions, the dissolved ortho-phosphate appears to behave essentially as a conservative substance.

Some production is apparent in the reaches above the Sand as evidenced by the increasing chlorophyll A level. However, below that confluence the chlorophyll A value remains essentially stable between 10 and 12 µg/l, indicating little additional fixation. The activity of the organisms is further indicated by the diurnal variation in dissolved oxygen at two stations on the mainstem (Figure 5.2). This diurnal variation is apparently due to the activity of attached growth on the bed and planktonic species being transported with the water. Since phosphorous can be adequately modelled as a conservative substance and despite the evidence of active photosynthesis, it appears that nutrient uptake from the system equals the input from external sources.

Ammonia levels measured during the first survey were below the limits of detection (0.02 mg/L). Ammonia levels in the second survey were measurable but were again associated with very low nitrate levels. Once again phosphates behaved essentially conservatively, further indicating a low level of nutrient uptake in the mainstem.

Given these low levels, the calibrations of nitrogen species and algae were not carried out. The results of the calibration of dissolved ortho-phosphate is shown in Figures 5-3 to 5-5.

5.5 Biochemical Oxygen Demand (BOD)

The data upstream of the confluence with the Sand indicate a constant BOD concentration of 1.0 mg/l. This appears inconsistent with the agricultural landuse in the area and cannot be adequately explained using the data collected during the sampling survey. In this area the results are adequately simulated by a BOD decay rate (K_1, base 'e') of 0.25 day^{-1} and an incremental input of 5 mg/l. This decay rate is within the range normally cited for K_1 (0.1 to 2, Tetra Tech, 1978). Table 5.7 shows the calibrated BOD decay rates for the Beaver River system.

Below the confluence of the Sand the instream BOD concentration increased to as high as 3.2 mg/l at the border. Attempts to simulate this behaviour through adjustment of the incremental inflow concentrations and BOD decay rate were reasonably successful. In Figure 5-6, the line $K_1 = K_c$ shows the calibrated value. A factor to be considered when viewing these data is the uncertainty of the BOD test in this low range. Standard Methods (1978) indicates that since laboratory conditions cannot duplicate natural conditions and because of the importance of sample handling, the BOD test may not be well suited to the analysis of surface waters. It may be said of the Beaver River, however, that very little BOD was present at the time of sampling.

Marie Creek receives the discharge from the CFB-Cold Lake sewage treatment plant. Data on the current quality of the discharge was unavailable so 1970 values were investigated. These were found to be indicative of upset conditions and were not used. Typical values for a moderately well operated plant were used. The results are shown in Figure 5-7.

Muriel Creek exhibits high values of BOD, as it does other parameters. The results of the calibration are shown in Figure 5-8.

Also in Figures 5-6 to 5-8, the results of sensitivity analysis carried out on BOD concentrations are shown. The results shown illustrate the effect of reducing the calibrated decay

TABLE 5.7

CALIBRATED BOD AND COLIFORM DECAY RATES

Reach Order	Reach Identification	BOD Decay Rate, k (day^{-1})	Total Coliform Dieoff Rate (day^{-1})
1.0	Beaver 1	0.13	0.53
2.0	Beaver 2	0.13	0.30
3.0	Beaver 3	0.13	0.53
4.0	Beaver 4	0.13	0.53
5.0	Mooselake 1	0.25	0.00
6.0	Beaver 5	0.13	0.53
7.0	Sand 1	0.10	0.00
8.0	Sand 2	0.10	0.00
9.0	Sand 3	0.10	0.00
10.0	Sand 4	0.10	0.00
11.0	Sand 5	0.10	0.00
12.0	Sand 6	0.10	0.00
13.0	Sand 7	0.10	0.00
14.0	Sand 8	0.10	0.00
15.0	Sand 9	0.10	0.00
16.0	Beaver 6	0.05	0.53
17.0	Manatokan 1	0.25	0.53
18.0	Beaver 7	0.05	0.53
19.0	Beaver 8	0.05	0.53
20.0	Beaver 9	0.05	0.53
21.0	Jackfish 1	0.25	0.53
22.0	Jackfish 2	0.25	0.53
23.0	Jackfish 3	0.25	0.53
24.0	Jackfish 4	0.25	0.53
25.0	Beaver 10	0.05	0.00
26.0	Beaver 11	0.05	0.53
27.0	Beaver 12	0.05	0.53
28.0	Marie 1	0.25	0.53
28.1	Marie 1.1	0.40	0.50
29.0	Marie 2	0.40	0.50
30.0	Beaver 13	0.05	0.00
31.0	Muriel 1	0.43	0.53
32.0	Muriel 2	0.43	0.53
33.0	Muriel 3	0.43	0.53
34.0	Muriel 4	0.43	0.53
35.0	Muriel 5	0.20	0.53
36.0	Beaver 14	0.05	0.53
37.0	Beaver 15	0.05	0.53
38.0	Beaver 16	0.05	0.53

rate by 50 per cent (K_1 = .5 K_1C) and increasing it by 200 per cent (K_1 = 2 K_1C). The doubling of K_1 is seen to have a large relative affect upon the BOD in the Beaver River. However, this was not reflected in a change in the dissolved oxygen regime of the main stem.

5.6 Dissolved Oxygen

The selection of a suitable reaeration coefficient (K_2) is a very important part of the modelling process. Many investigations have developed empirical formulae relating stream characteristics such as flow, turbulance, surface area, velocity and depth to field measured values of K_2. None of these relationships are universally applicable and so the judicious choice of the most appropriate form is required. QUAL II provides seven options by which to calculate K_2 empirically, plus the option to read in field measured values.

The selection of a relationship for K_2 was carried out using a graphical method proposed by Covar (1976). This method showed that K_2 relationships developed by O'Connor and Dobbins (1958), Churchill et al. (1962) and Owens and Gibbs (1964) can be applied to three distinct areas on a log-log plot of depth vs velocity. Using this method the O'Connor and Dobbins relationship was applied to the Beaver and Sand Rivers and the Owens and Gibbs relationship for the small tributaries.

Figures 5-9 to 5-11 show the results of the calibration. The mainstem and the tributaries are essentially saturated with dissolved oxygen. Variation from the saturation value is attributed to the activity of aquatic plants and algae during the sampling survey. This variation is particularly pronounced in Marie Creek which shows a concentration of 4.5 mg/l below the saturation value. Such a decrease may cause stress upon aquatic vertebrates in the system and should be observed carefully. In the steady state mode Qual II does not simulate the diurnal dissolved oxygen cycle. Accordingly, information on the amplitude of the diurnal variation must be acquired at critical reaches so that model predictions can be interpreted correctly. Figure 5-2 shows that the diurnal dissolved oxygen variation can be as high as 2 mg/L. Accordingly, since the model predicts daily average dissolved oxygen these should be corrected by approximately half the observed diurnal variation, to estimate critical concentrations.

Also shown on the figures are the results of using other empirical formulations to calculate K_2 on the Beaver River. This analyses indicated that the dissolved oxygen response of the Beaver River was insensitive to varying K_2 formulations. This was due to the natural convergence of all methods at the saturation concentration.

Sensitivity analysis on the affect of BOD decay rate was not rigorously carried out. The oxygen consumed by the low mass of oxygen demanding substances in the river, even at a high rate, would be overwhelmed by the effect of reaeration.

5.7 Coliforms

In the Beaver River the concentration of total coliforms varied from 164/100 ml at Goodridge to 13 at the Saskatchewan border. These levels are well within the Alberta Surface Water Quality Objectives (1977).

The slightly elevated levels above the Sand confluence are probably due to runoff from agricultural landuse at the immediate edge of the river channel. An input level in incremental runoff of 150/100 ml was used in the upper reaches.

This corresponded to the value observed in Jackfish Creek and was, therefore, considered representative of inflow to the river. In the lower reaches the channel banks are not as heavily utilized and runoff is less contaminated. This results in a steadily decreasing instream concentration under the influence of dilution and die-off. In the upper reaches a die-off rate of 0.53 day was found to adequately fit the data observed (Figure 5-12, $K_5 = K_5C$).

Marie Creek exhibited slightly higher levels of total coliforms than did the mainstem (Figure 5-13, $K_5 = K_5C$). Despite the presence of a treated sewage outfall in the Medley area, the instream concentration did not increase appreciably downstream. This may be due to a very high instream die-off rate. However, the data is inadequate to identify this in detail. A die-off rate of 0.53 day^{-1} was used to fit the data combined with an incremental inflow value of 250/100 ml.

Muriel Creek exhibited the highest levels of total coliform bacteria (Figure 5-14, $K_5 = K_5C$). The high values originate at Charlotte Lake. Calibrated incremental inflows amounted to 200/100 ml and the same die-off rate of 0.53 day^{-1} was used. Despite the high values observed in Muriel Creek provincial water quality objectives were not exceeded.

Table 5.7 shows the calibrated total coliform decay rates. Figures 5-12 to 5-14 show the sensitivity of total coliform concentrations to varying die-off rates. At high total coliform levels, as shown in Figure 5-13, a variation of 67 per cent can be seen between the predictions of an extreme K_5 values (K_5 = 2 K_5C and K_5 = 0.5 K_5C).

5.8 Conservative Substances

Conservative constituents neither accumulate nor decrease in streams. Therefore, in the QUAL-II model, the source-sink term is a simple identity function relating the instream concentration only to the inputs and withdrawals. Three parameters were modelled as conservative substances, chloride, sulphate and sodium.

All three parameters exhibit similar behaviour in the Beaver River system. In the Beaver River itself, the agricultural activity above the Sand River contributed to slightly elevated levels of these conservative substances in those reaches. The diluting effect of the Sand reduced the levels of sodium and chloride to an essentially constant level of 11 mg/l and 2 mg/l respectively. Sulphate behaved more erratically below the Sand than did sodium or chloride.

The sulphate data collected during the first and second surveys is shown in Table 5.8. Both sets of data reveal a cyclical profile along the length of the mainstem. Mass balance calculations carried out using calibrated flow values along the length of the mainstem, indicated that that the decline in concentration between kilometres 69 and 54 represented a 37 per cent loss of sulphate mass from the river.

TABLE 5.8

BEAVER RIVER SULPHATE DATA

Station No.	Location km	Survey No. 1 mg/1	Survey No. 2 mg/1
B1	143	8.0	13
B2	135	12.0	25
B3	124	17.0	31
B4	116	17.0	45
B5	107	21.0	35
B9	101.5	----	33
B11	100	<5	10
B12	78	12	10/18
B13	69	13	13
B16	54	8.0	13/14
B17	40	8.0	14
B18	39	21.0	9
B28	28	17.0	18
B31	0.0	12.0	9

During the second sampling survey two samples were collected at stations B12 and B16. As shown in the table some variability was observed in the values reported. This variability may be due to sampling or analytical error and would help to account for the large changes in flux observed. Given this high variability, the sulphate calibration results and levels must be interpreted with caution.

In Marie Creek the influence of the CFB-Cold Lake sewage treatment plant discharge can be observed with the abrupt increases in the concentrations of all three parameters. Muriel Creek concentrations of conservatives are the highest in the basin (Na = 200 mg/l, SO_4 = 400 mg/l, Cl>50 mg/l). This is attributed to its course through a highly agricultural basin. Calibrated incremental runoff concentrations in the Muriel Creek basin are Na, 560 mg/l, SO_4, 760 mg/l and Cl, 560 mg/l.

The results of the calibrations are shown in Figures 5-15 to 5-23.

5.9 Phenol

The levels of phenol found during both sampling surveys were generally less than the national limits of 0.005 mg/L (Appendix A). Most values ranged between 0.002 and 0.005 mg/l. Slightly higher values were found in Muriel Creek.

Phenol was treated as a non-conservative substance in the model calibration. However, with levels approaching the limits of detection no attempt was made to apply a decay rate in the mainstem. Levels in Muriel Creek were significant however and a decay rate of 0.3 day^{-1} was selected. (Figures 5-24 to 5-26). A single measured value of 0.003 mg/L in Marie Creek could not be adequately simulated.

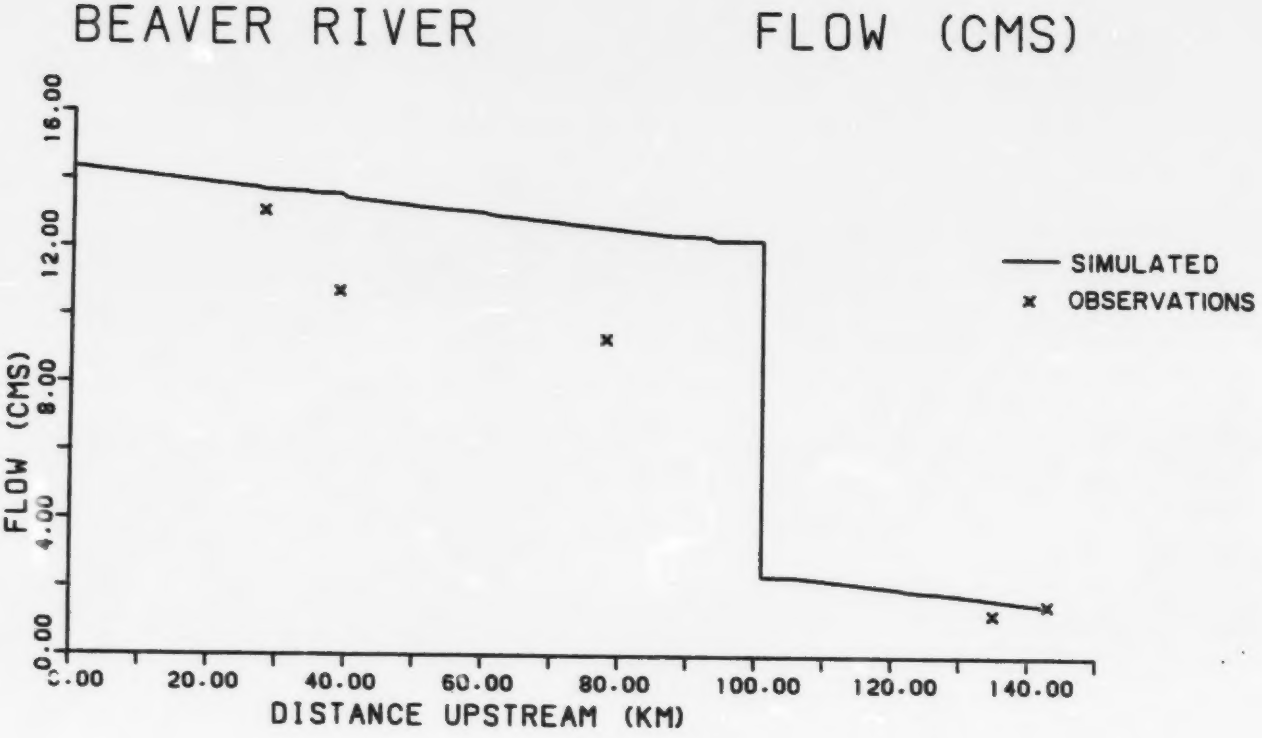


FIGURE 5.1 **BEAVER RIVER FLOW CALIBRATION**

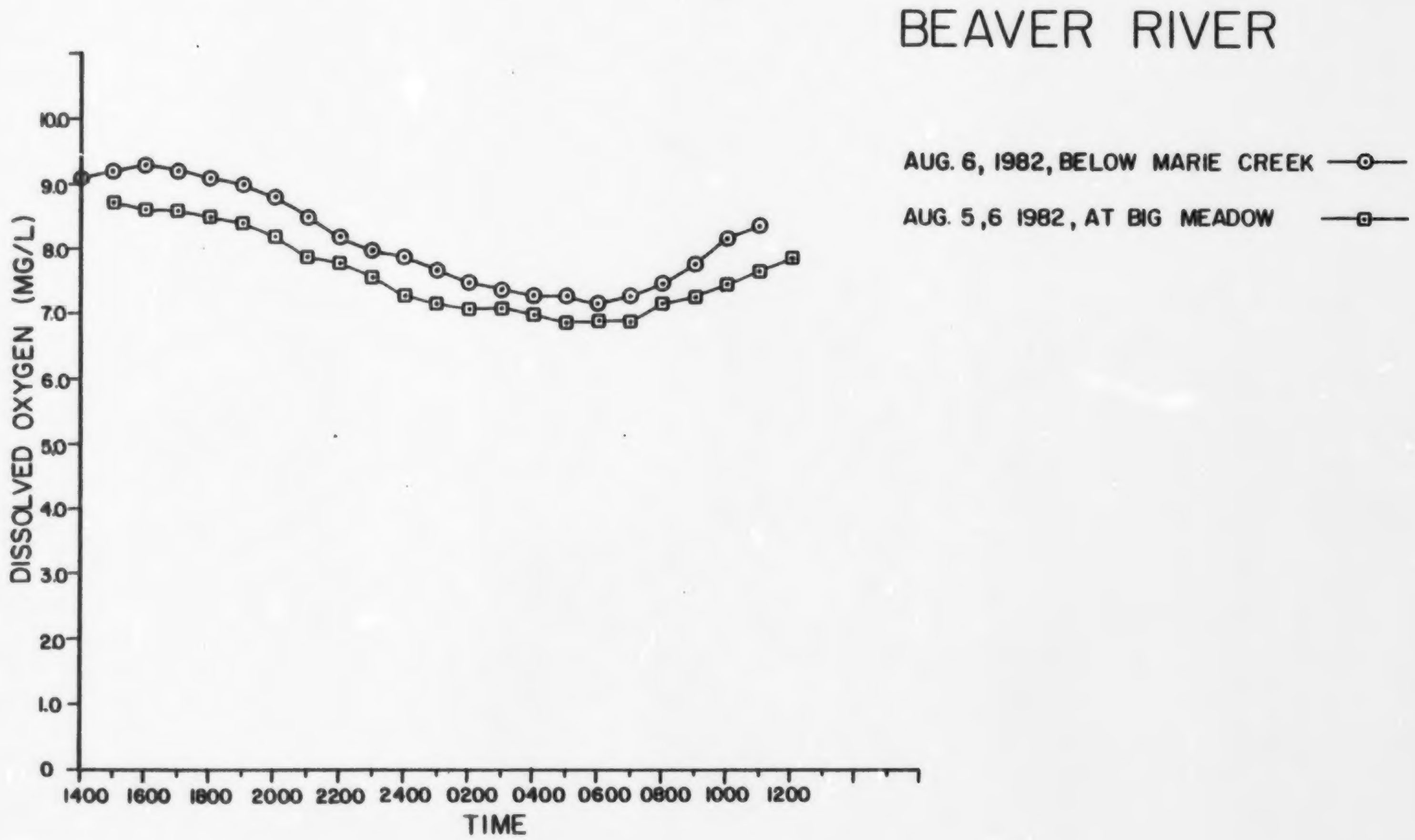


FIGURE 5.2 DIURNAL DISSOLVED OXYGEN VARIATION

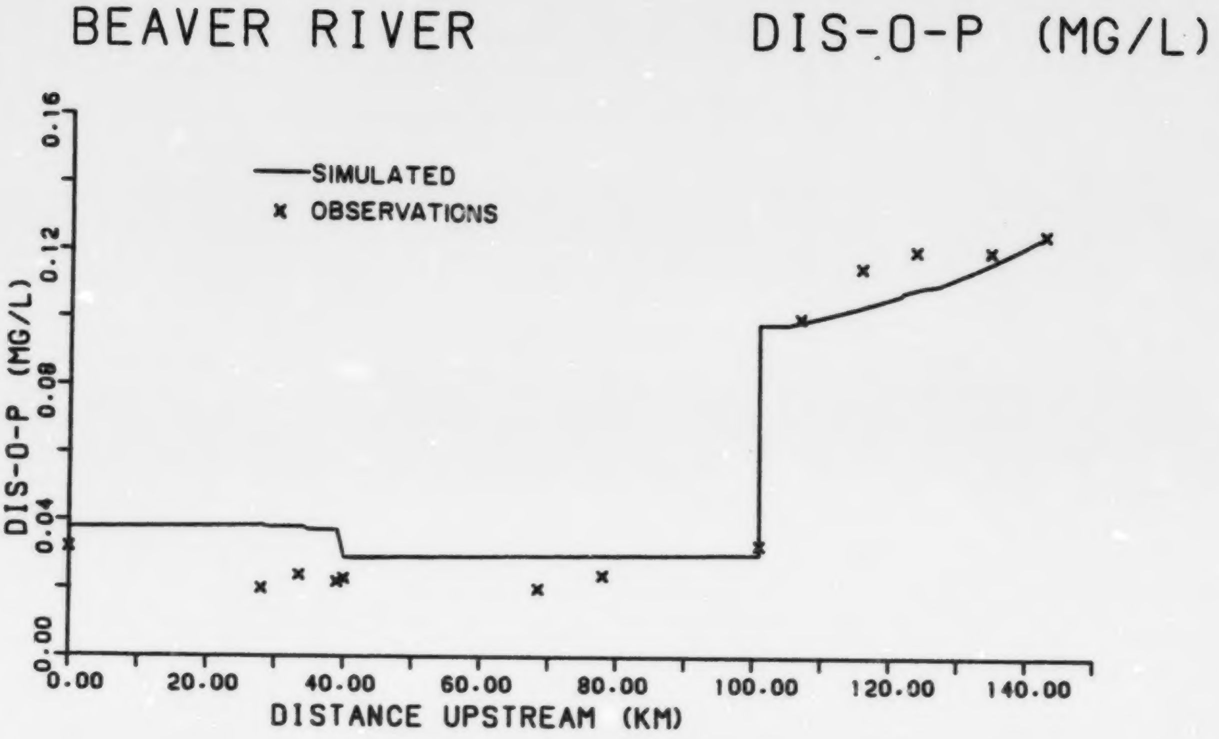


FIGURE 5.3 **BEAVER RIVER DISSOLVED ORTHO PHOSPHORUS CALIBRATION**

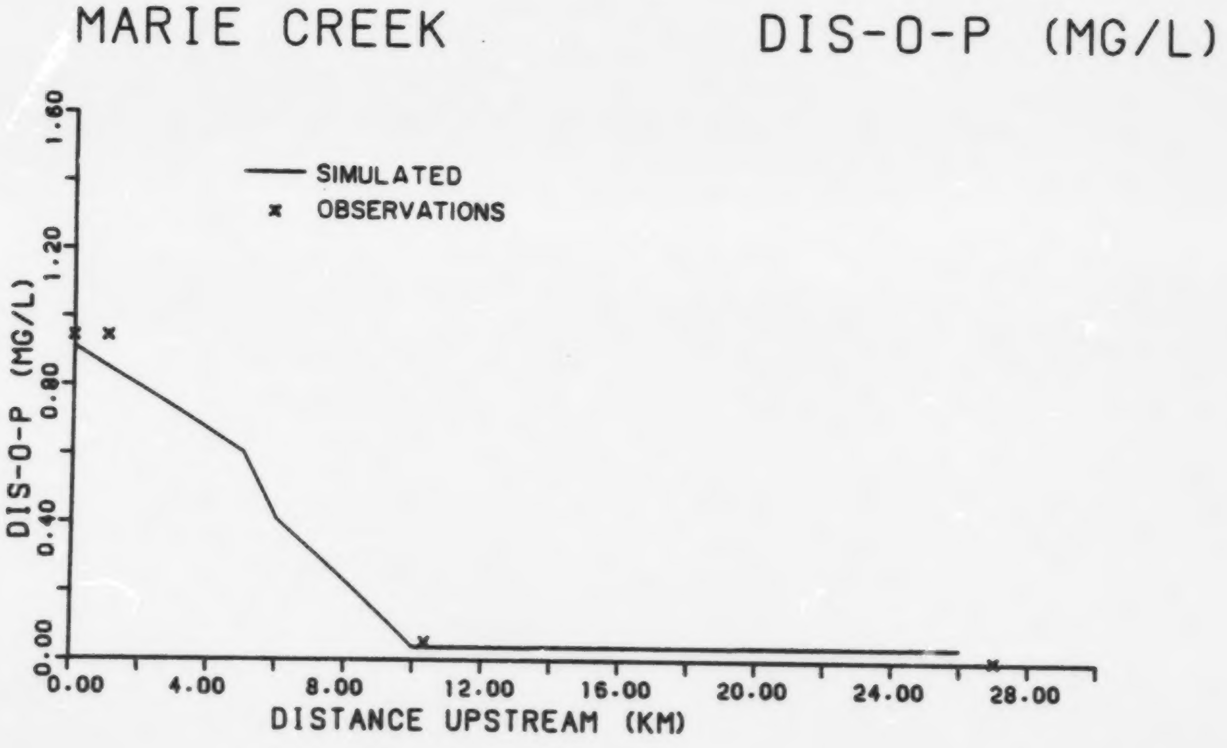


FIGURE 5.4 **MARIE CREEK DISSOLVED ORTHO PHOSPHORUS CALIBRATION**

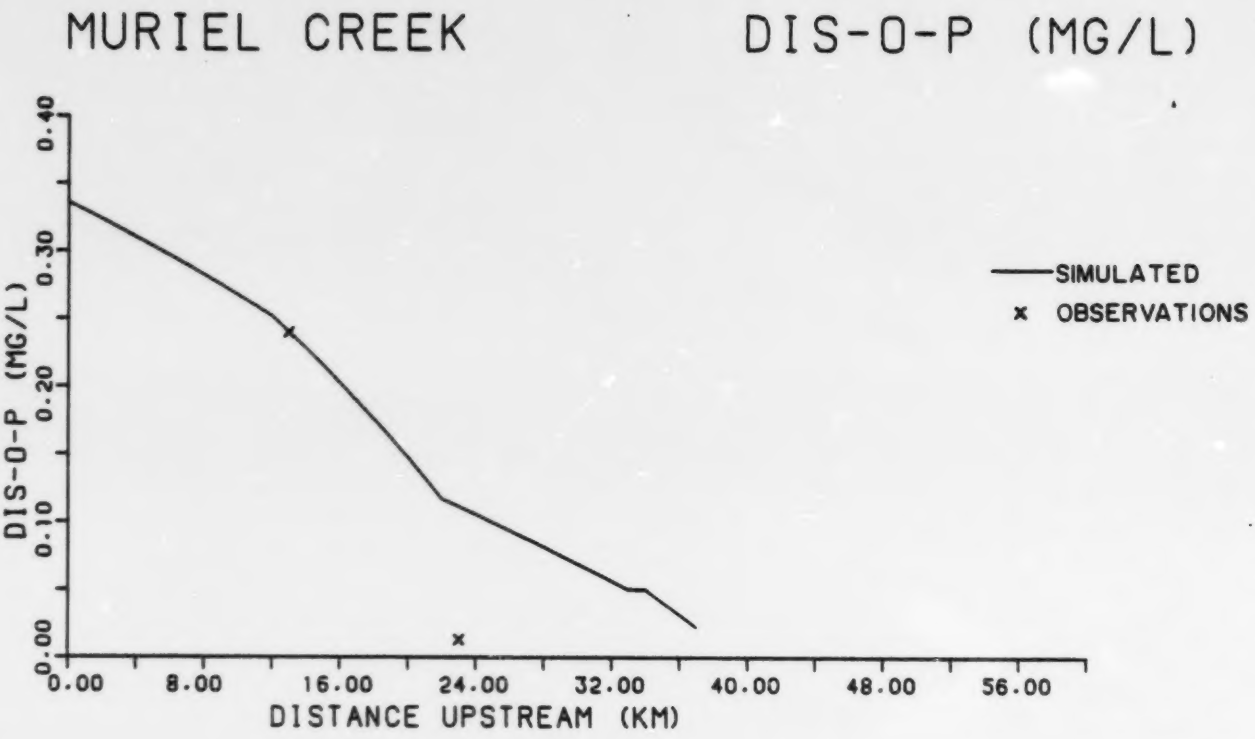


FIGURE 5.5 MURIEL CREEK DISSOLVED ORTHO PHOSPHORUS CALIBRATION

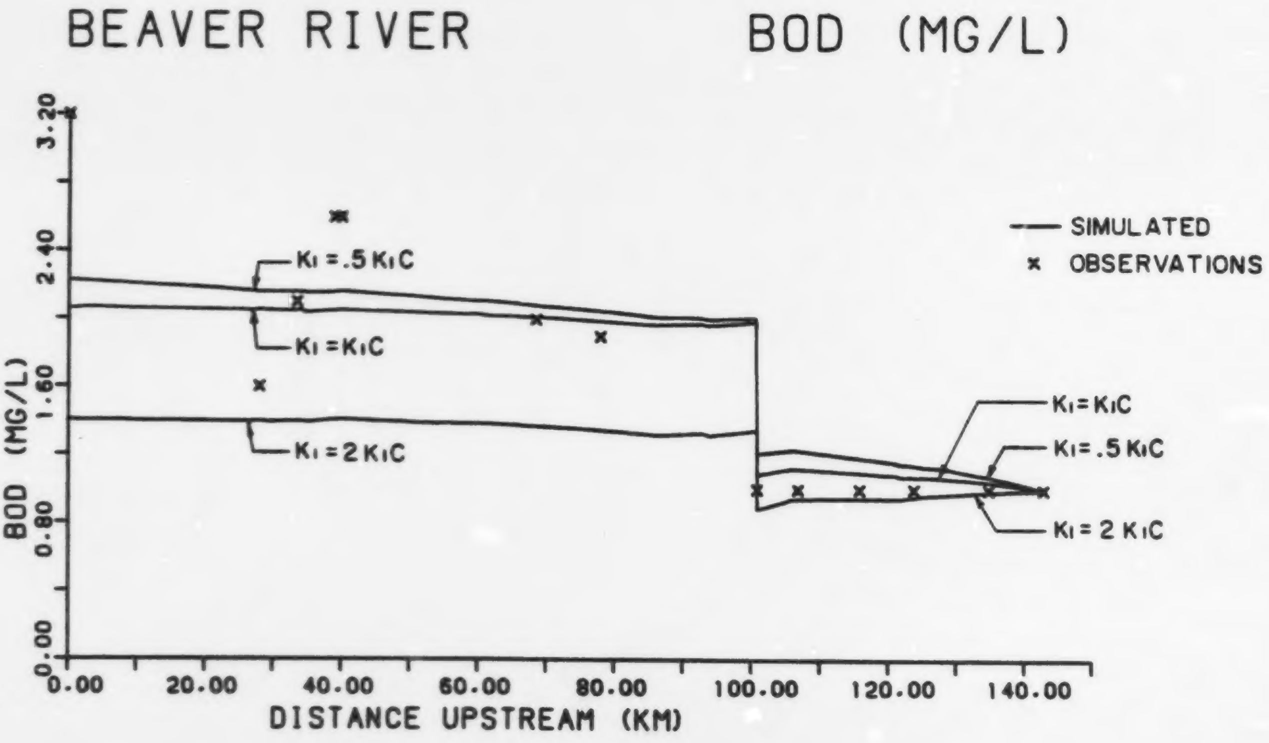


FIGURE 5.6 BEAVER RIVER BOD_5 CALIBRATION AND SENSITIVITY ANALYSIS

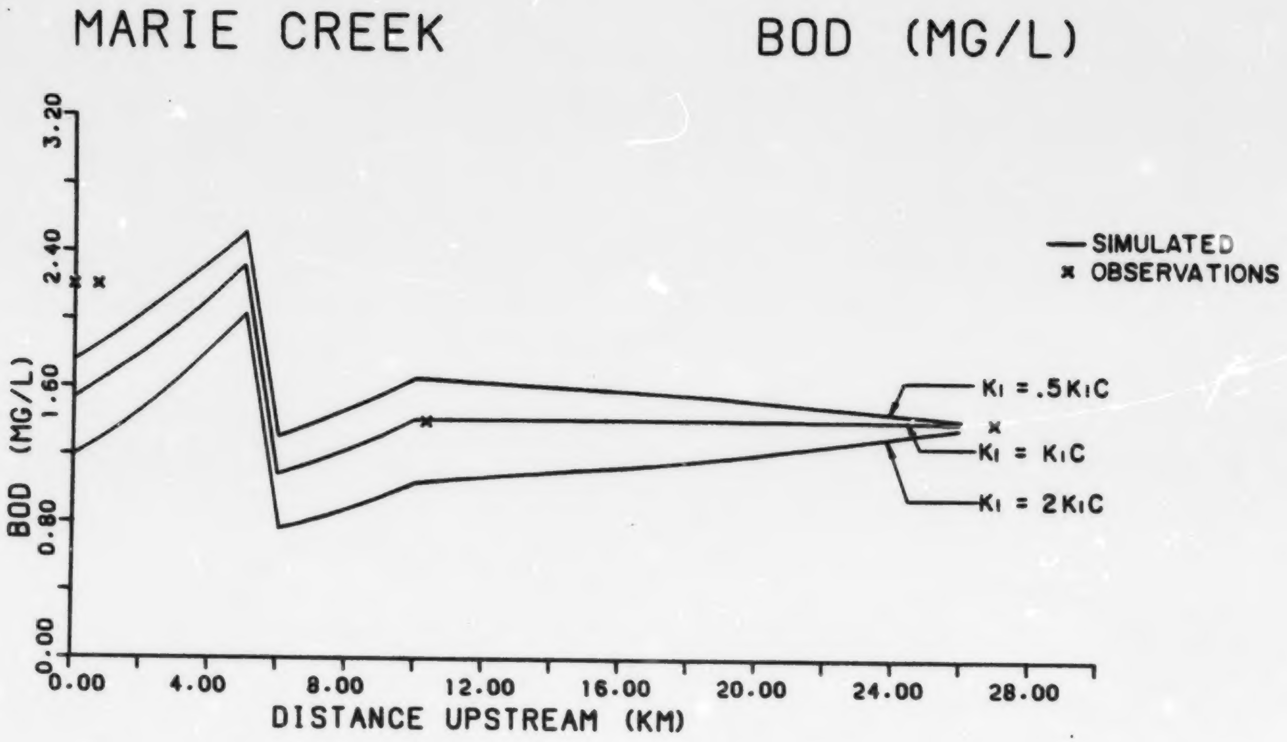


FIGURE 5.7 MARIE CREEK BOD_5 CALIBRATION AND SENSITIVITY ANALYSIS

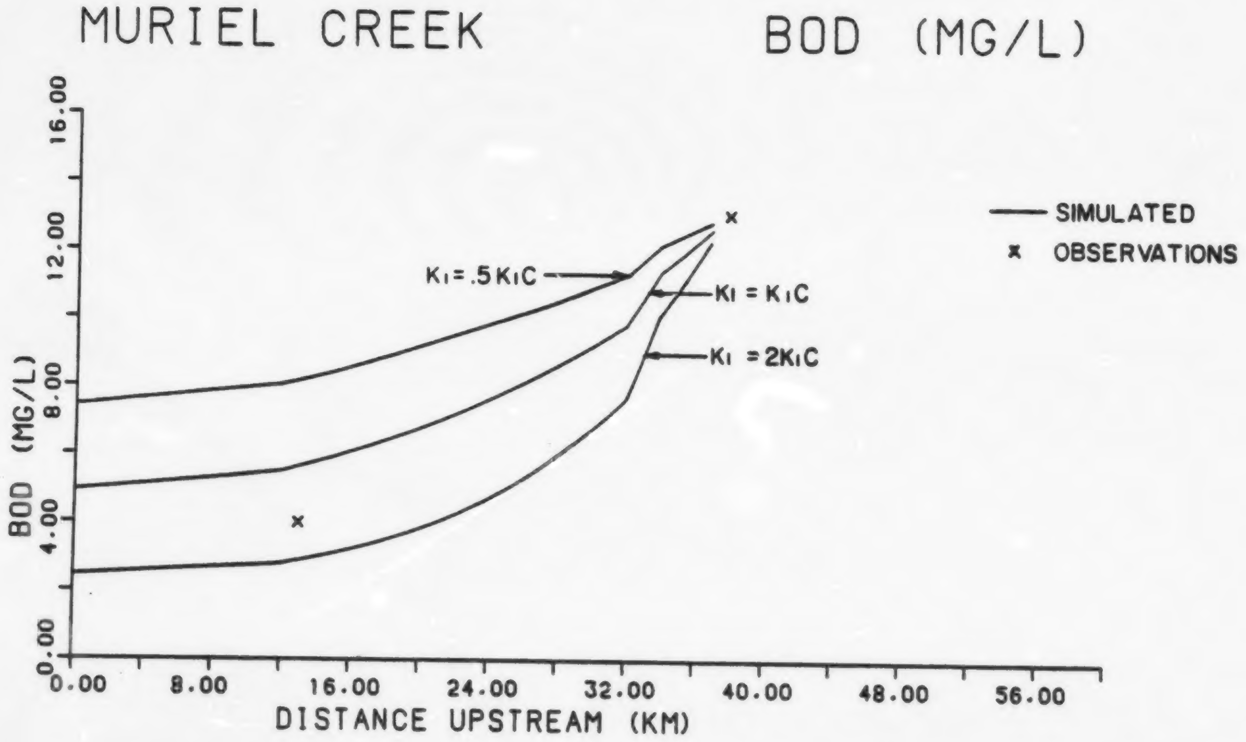


FIGURE 5.8 MURIEL CREEK BOD_5 CALIBRATION AND SENSITIVITY ANALYSIS

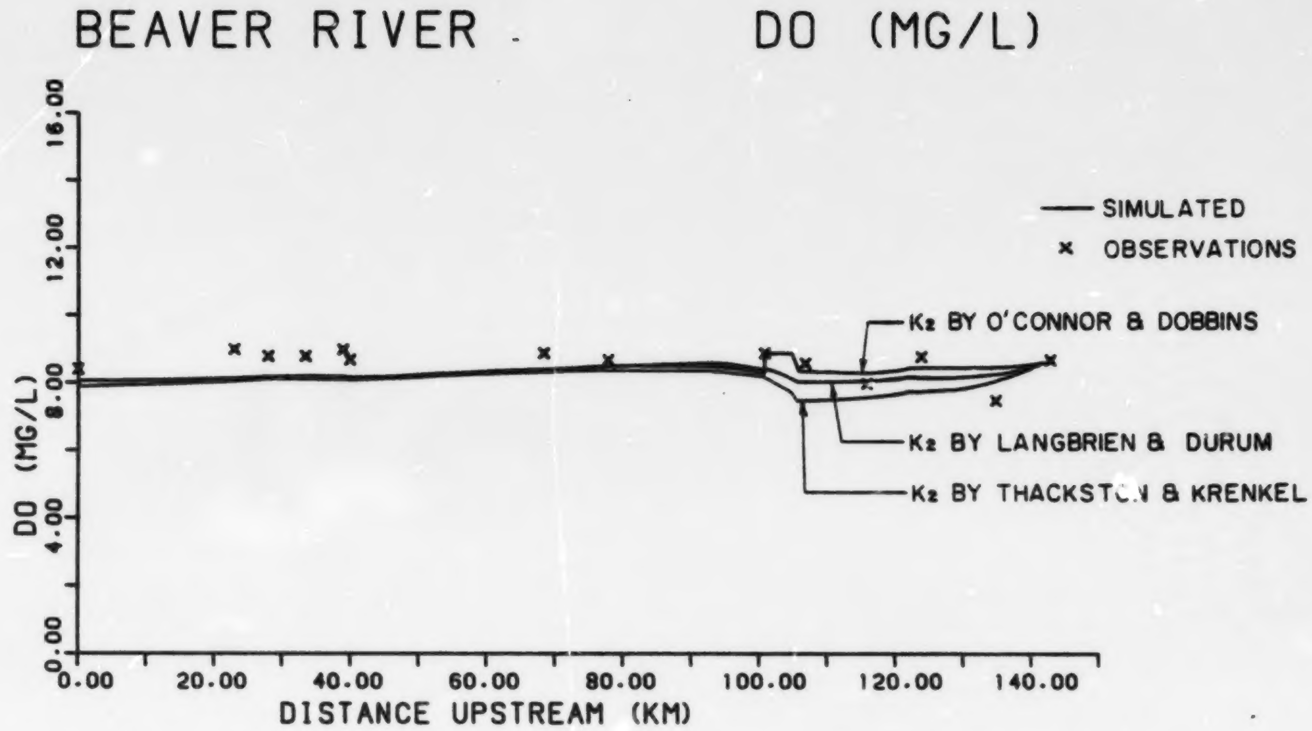


FIGURE 5.9 BEAVER RIVER DISSOLVED OXYGEN CALIBRATION AND SENSITIVITY ANALYSIS

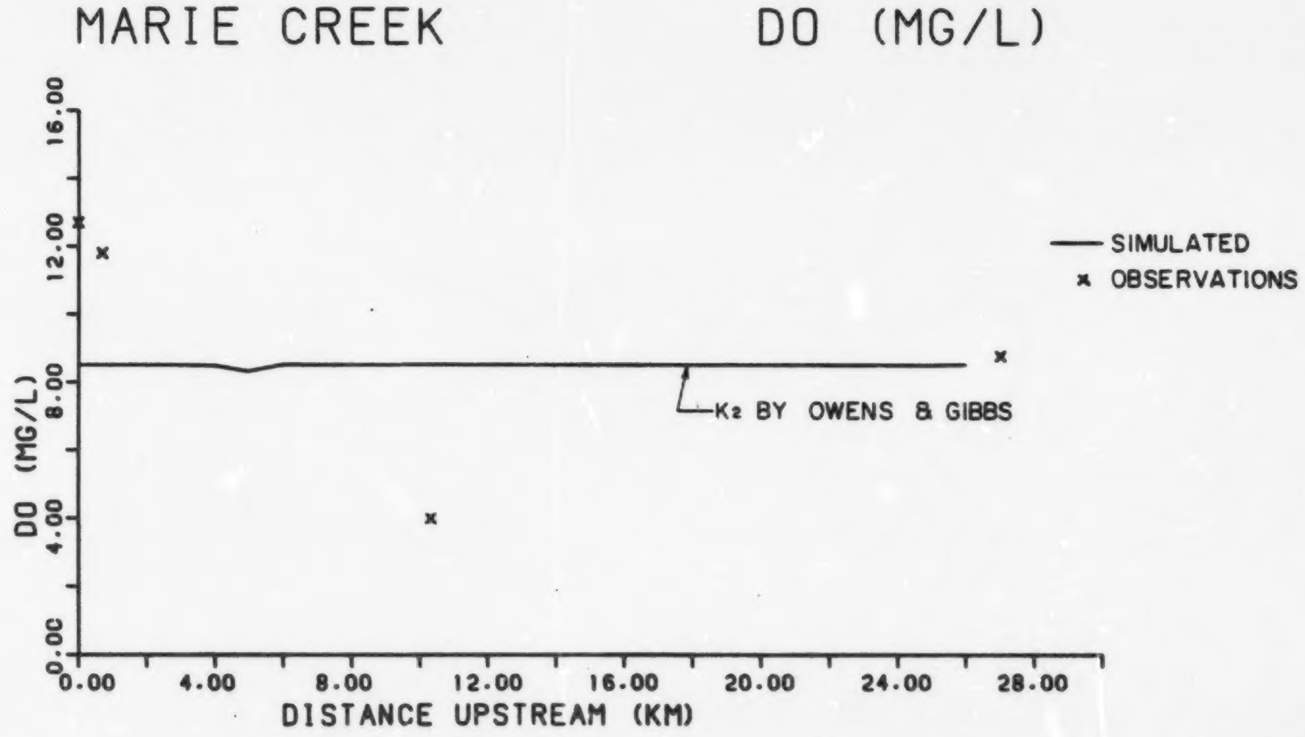


FIGURE 5.10 MARIE CREEK DISSOLVED OXYGEN CALIBRATION

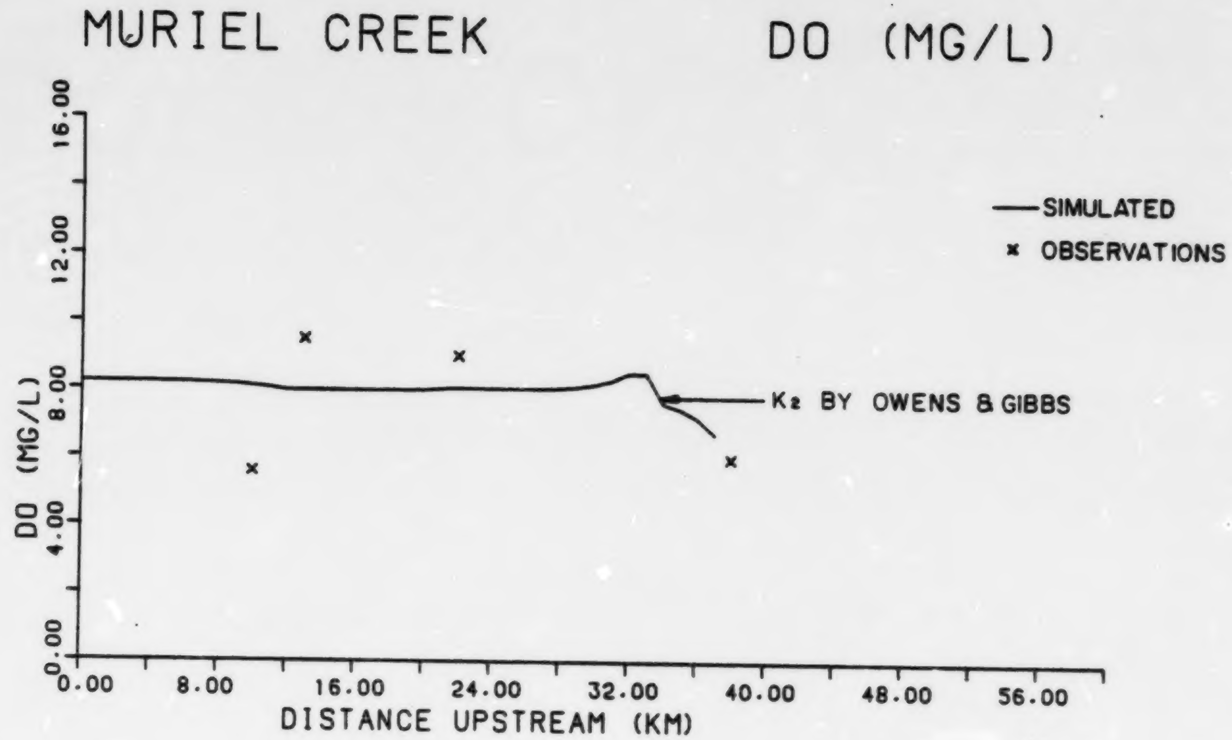


FIGURE 5.11 MURIEL CREEK DISSOLVED OXYGEN CALIBRATION

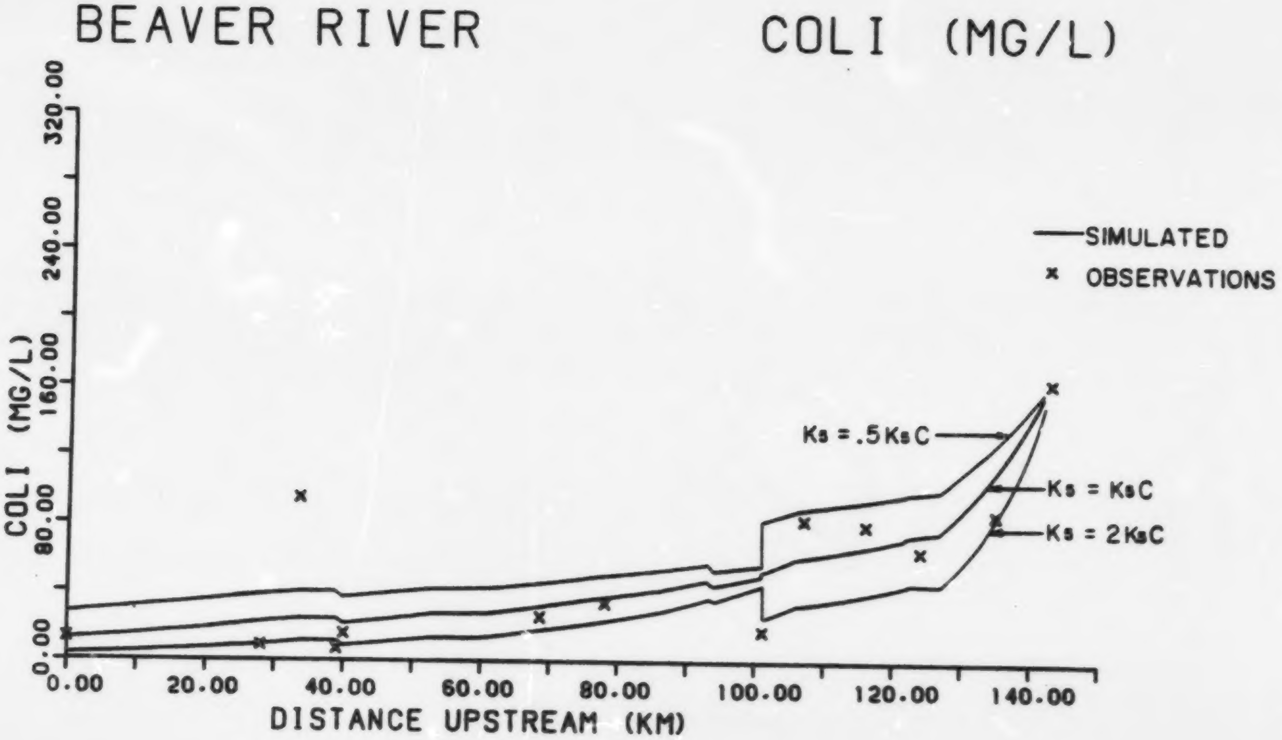


FIGURE 5.12 BEAVER RIVER TOTAL COLIFORM CALIBRATION AND SENSITIVITY ANALYSIS

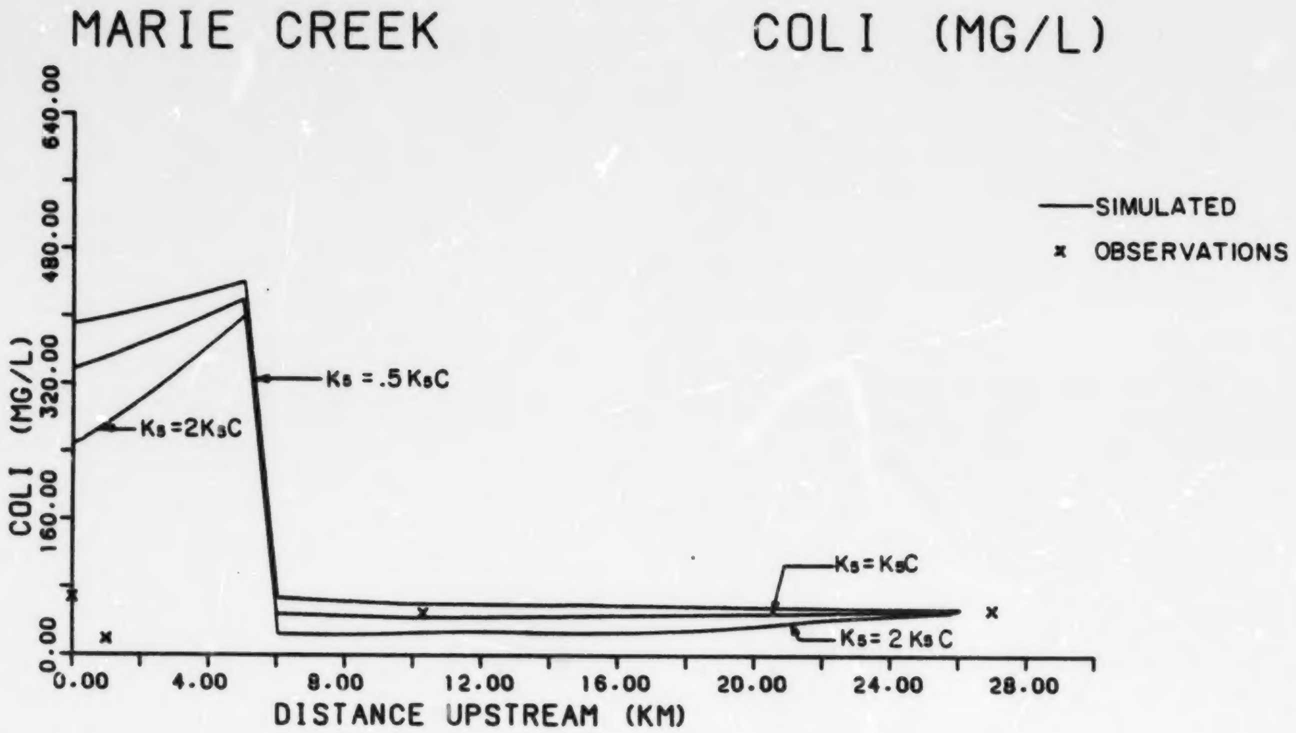


FIGURE 5.13 **MARIE CREEK TOTAL COLIFORM CALIBRATION AND SENSITIVITY ANALYSIS**

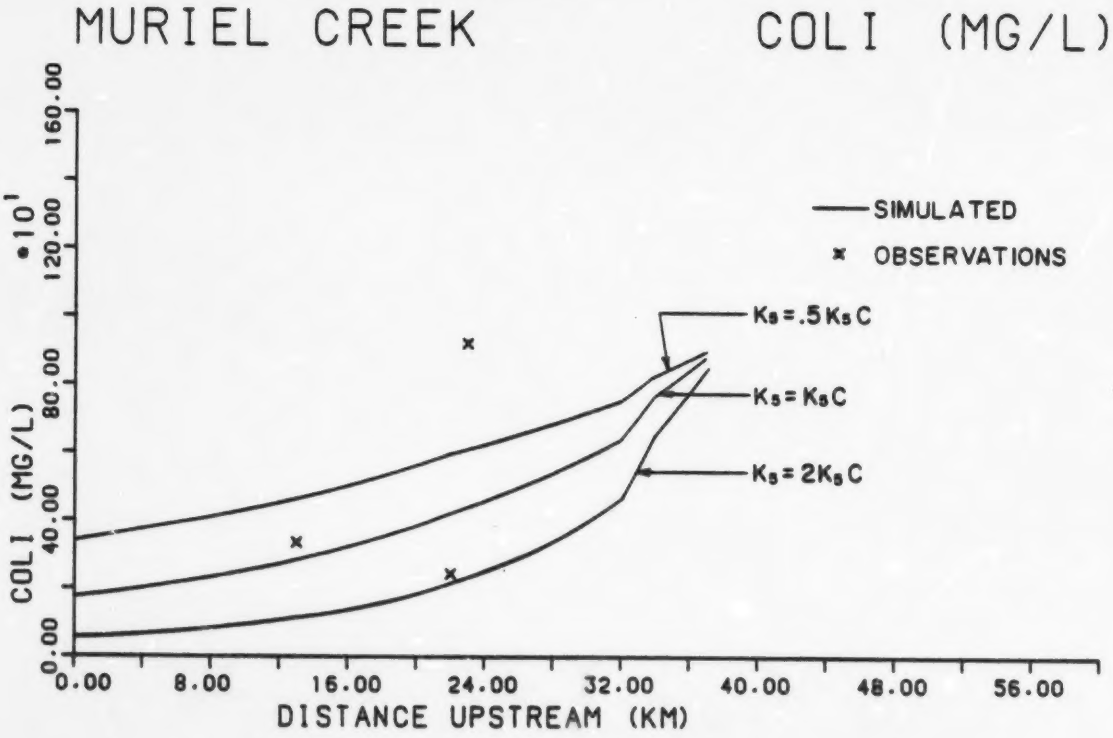


FIGURE 5.14 **MURIEL CREEK TOTAL COLIFORM CALIBRATION AND SENSITIVITY ANALYSIS**

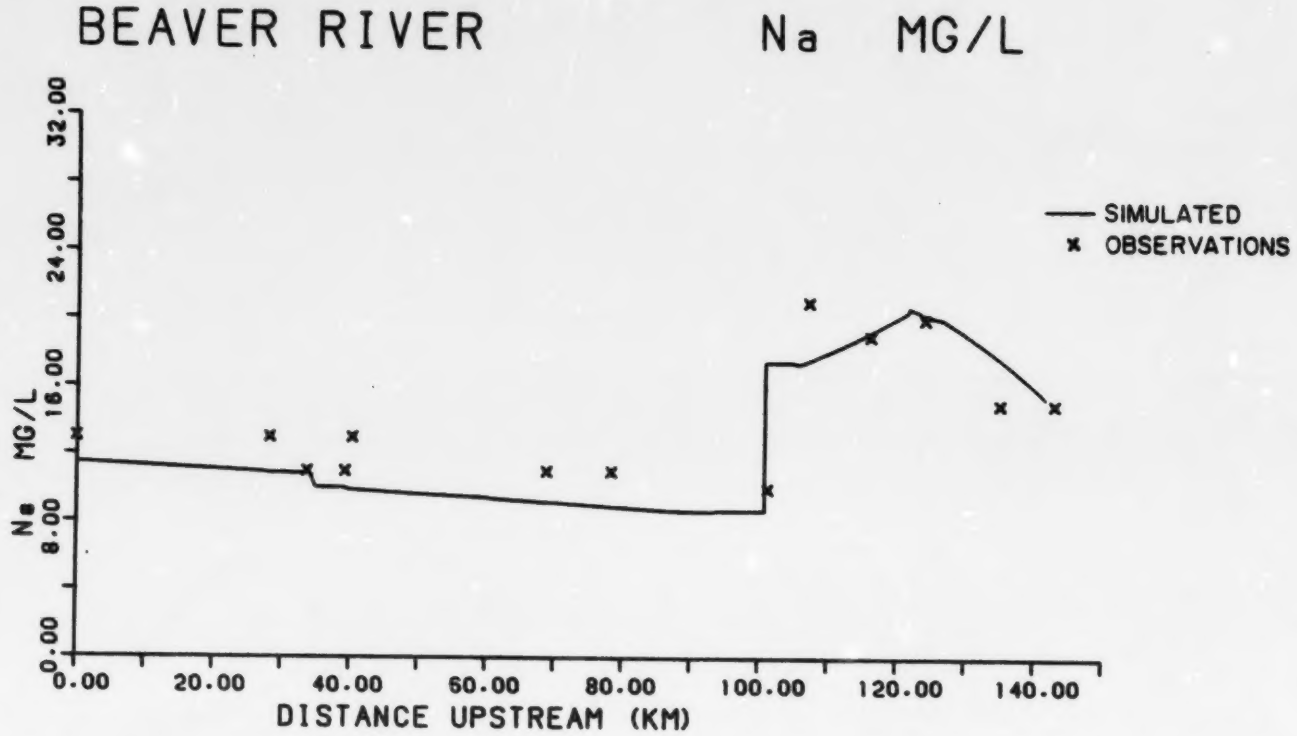


FIGURE 5.15 **BEAVER RIVER SODIUM CALIBRATION**

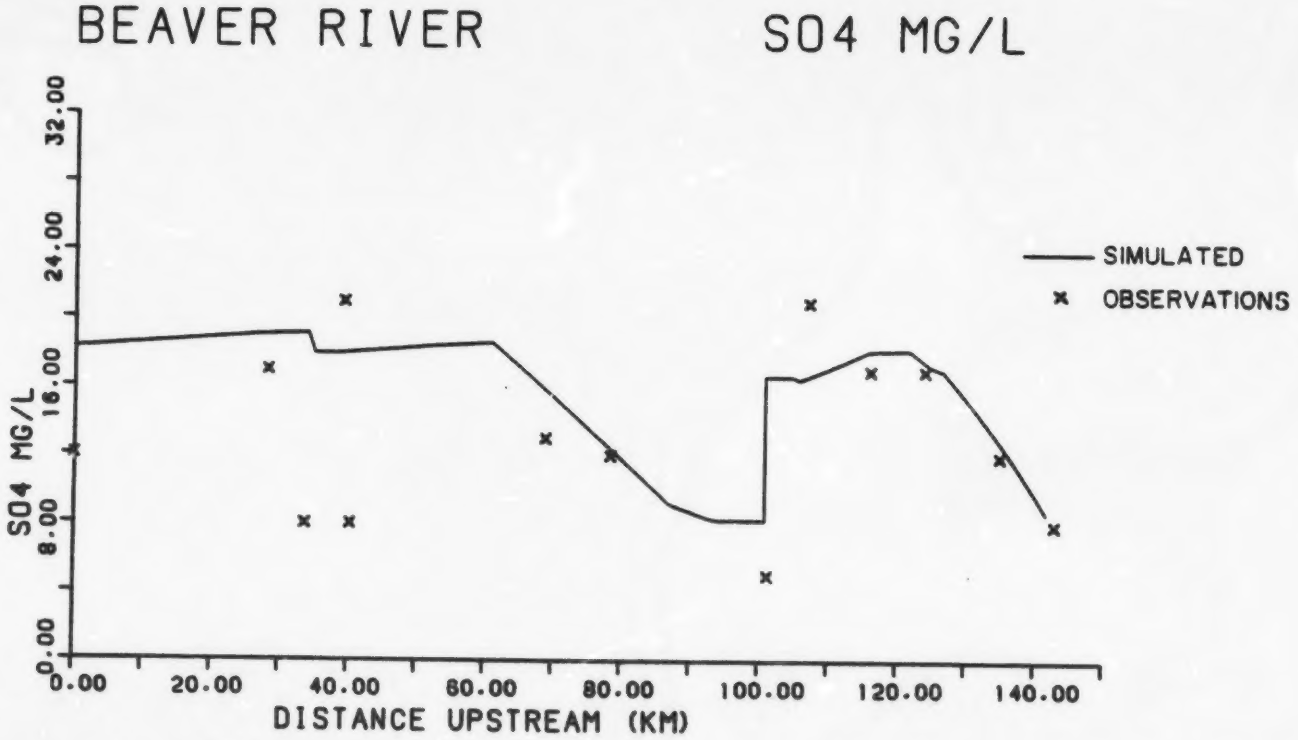


FIGURE 5.16 **BEAVER RIVER SULPHATE CALIBRATION**

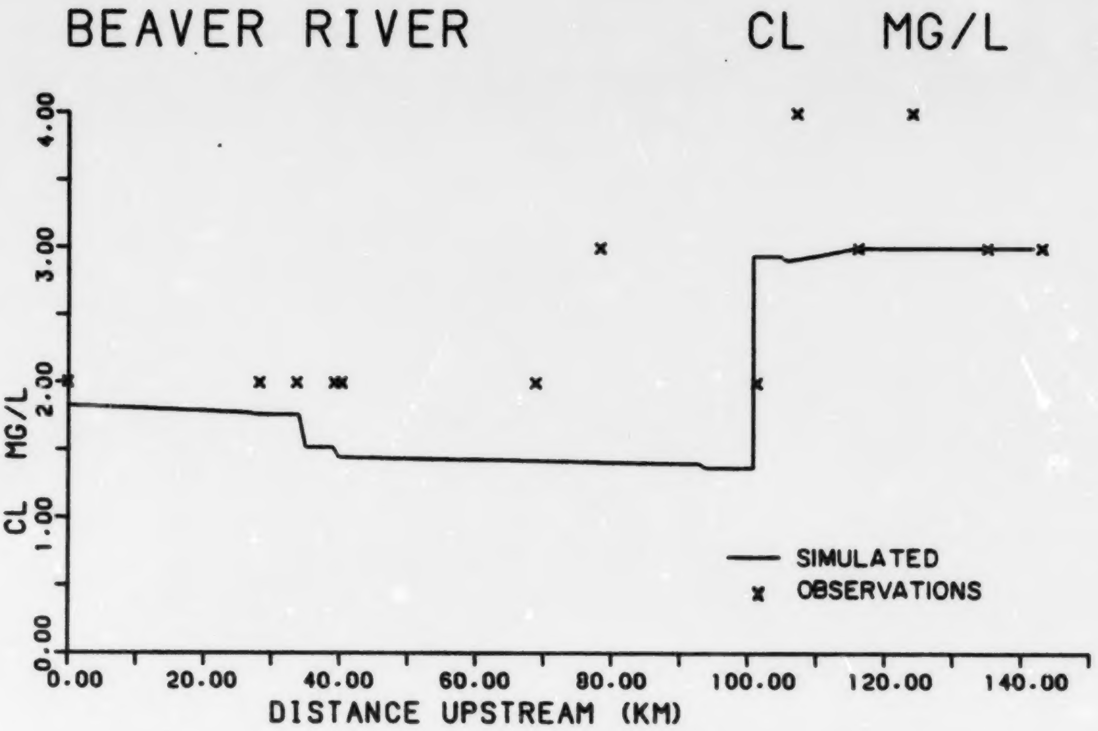


FIGURE 5.17 **BEAVER RIVER CHLORIDE CALIBRATION**

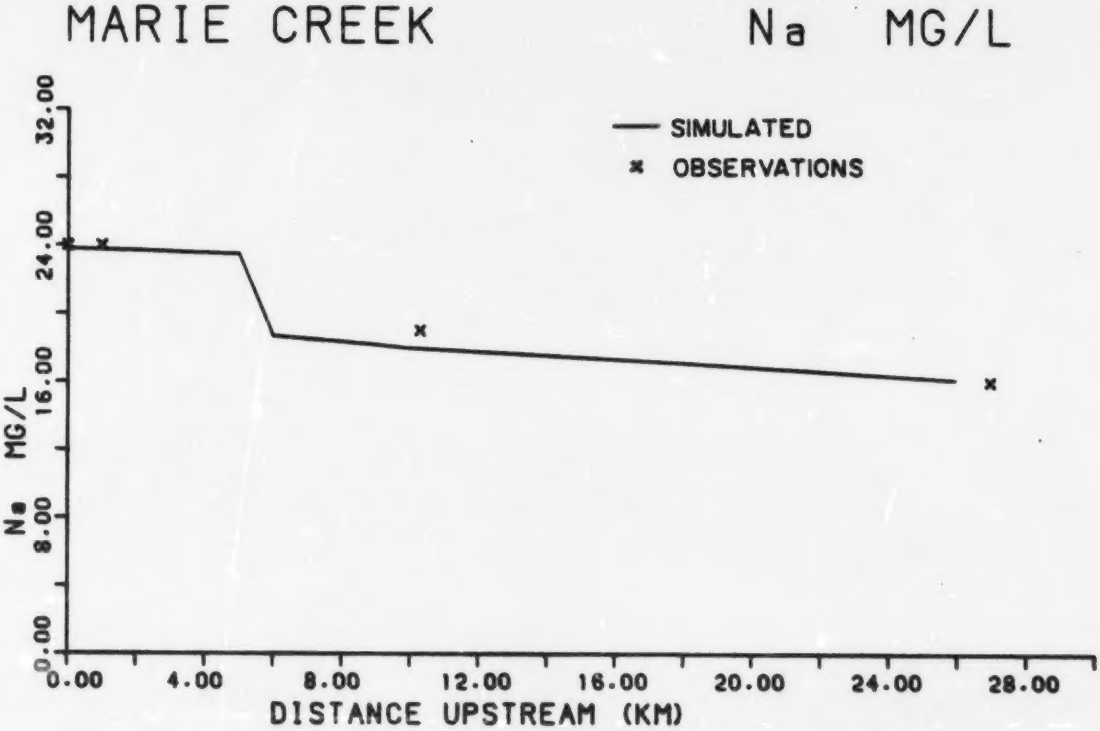


FIGURE 5.18 **MARIE CREEK SODIUM CALIBRATION**

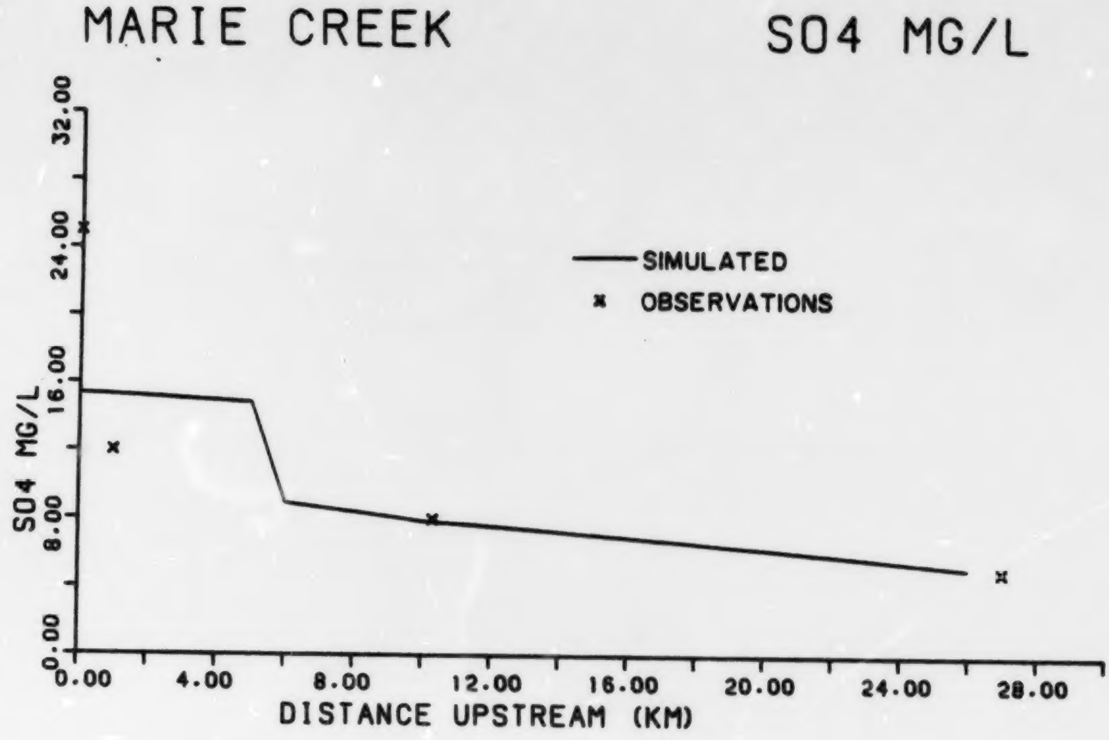


FIGURE 5.19 MARIE CREEK SULPHATE CALIBRATION

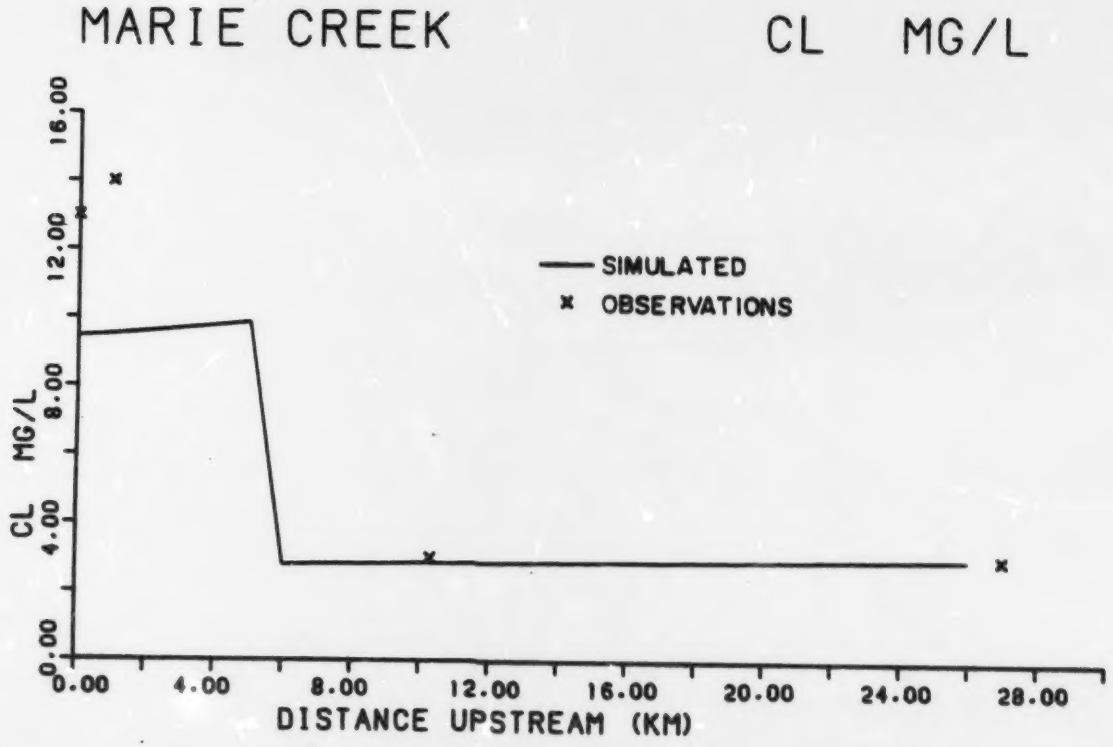


FIGURE 5.20 MARIE CREEK CHLORIDE CALIBRATION

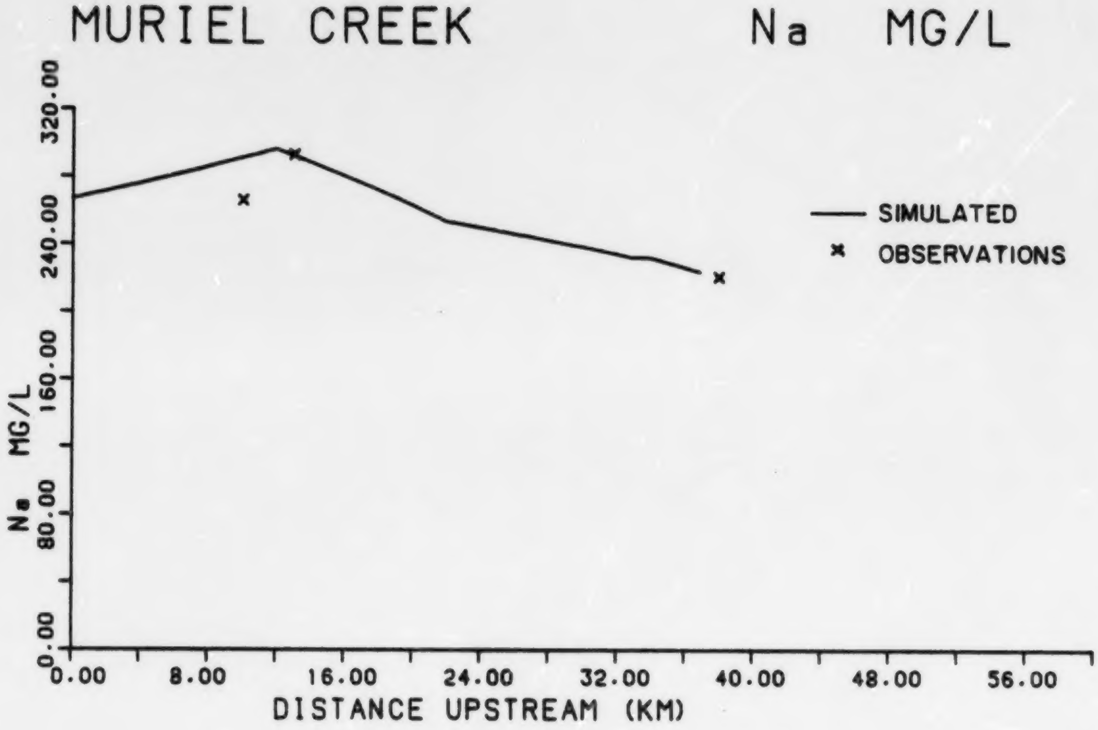


FIGURE 5.21 **MURIEL CREEK SODIUM CALIBRATION**

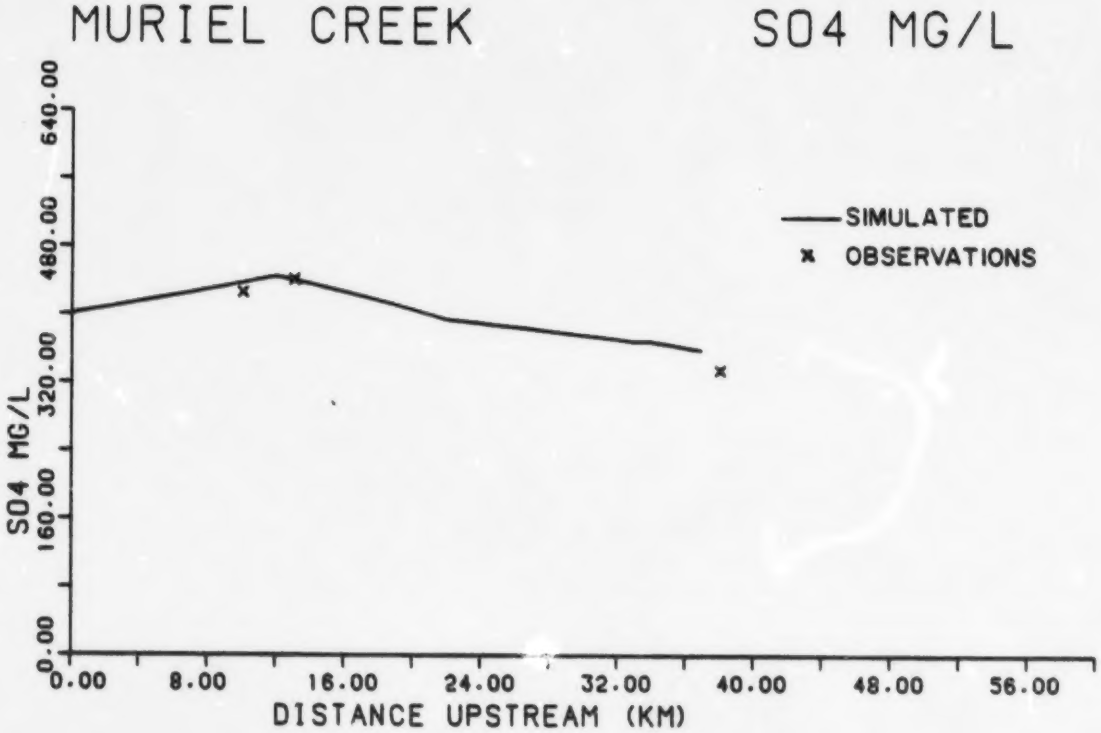


FIGURE 5.22 **MURIEL CREEK SULPHATE CALIBRATION**

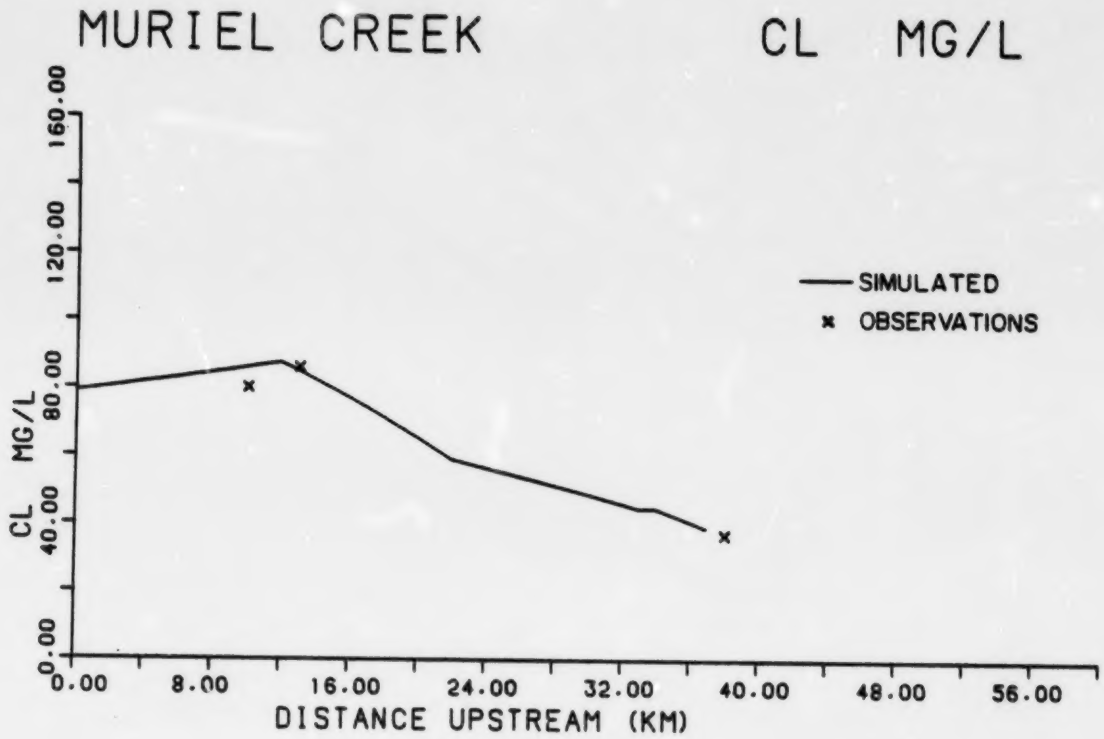


FIGURE 5.23 **MURIEL CREEK CHLORIDE CALIBRATION**

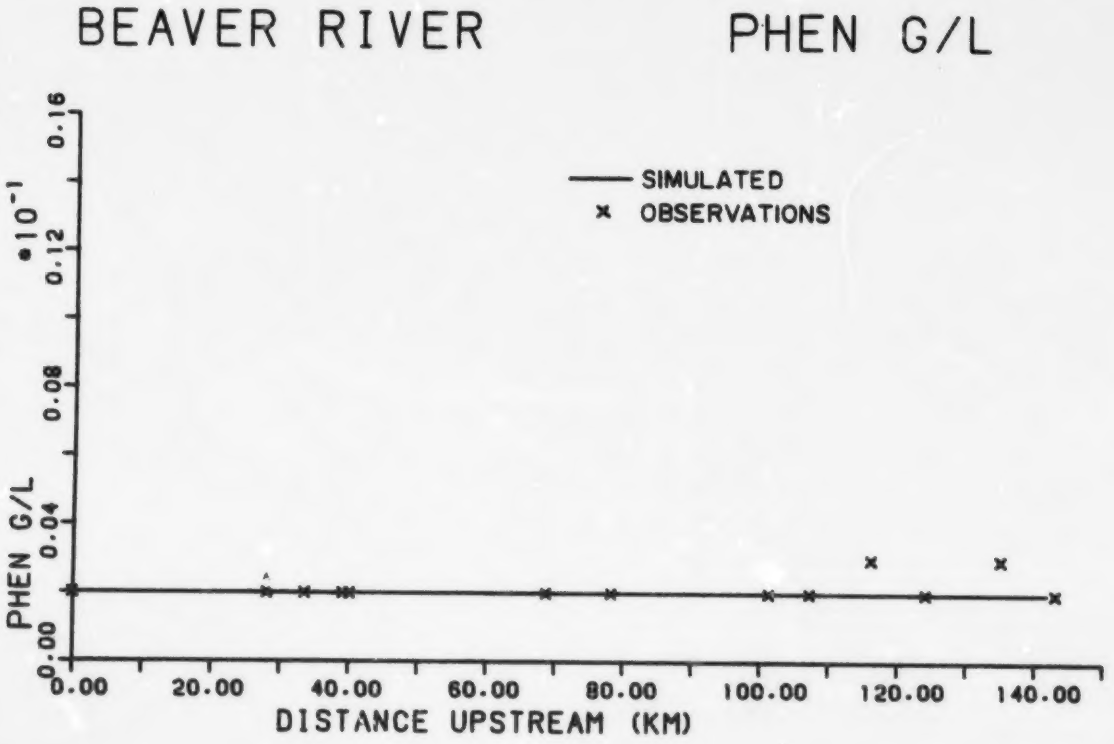


FIGURE 5.24 **BEAVER RIVER PHENOL CALIBRATION**

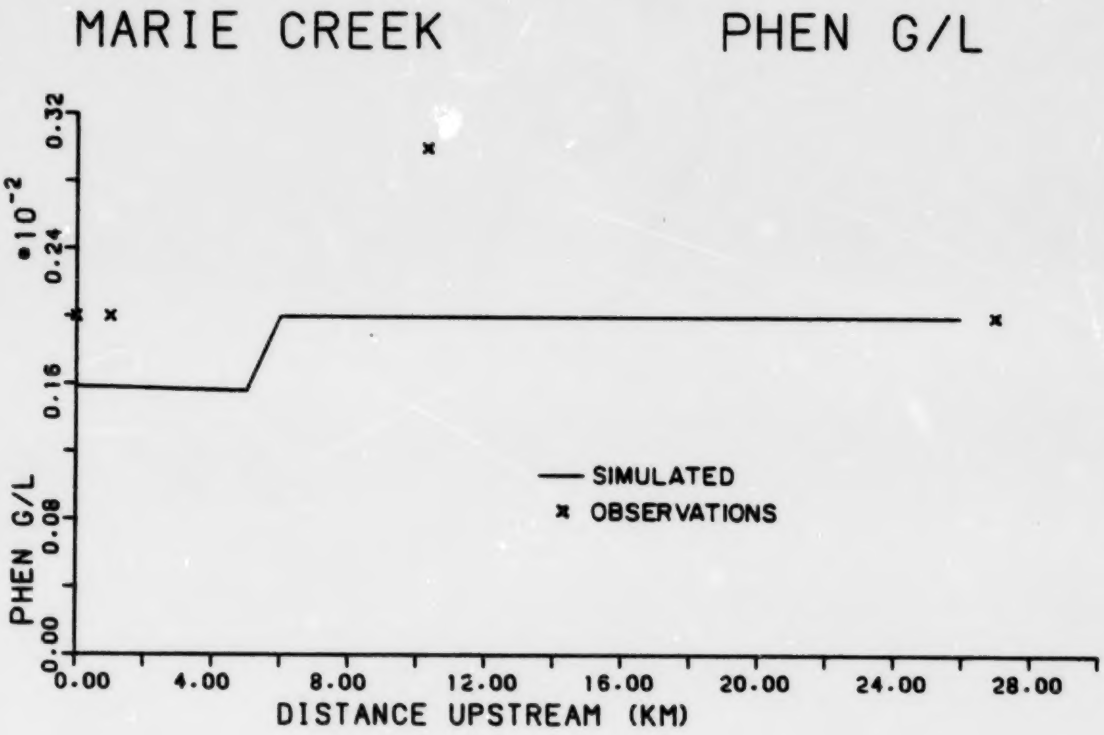


FIGURE 5.25 **MARIE CREEK PHENOL CALIBRATION**

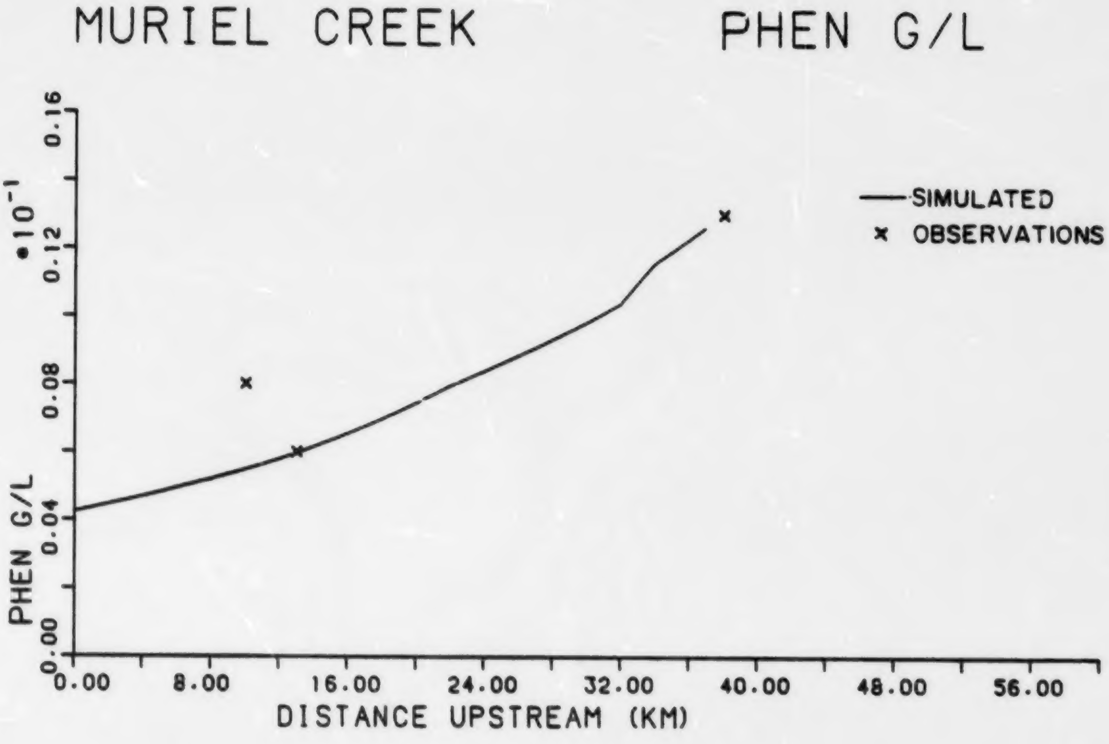


FIGURE 5.26 **MURIEL CREEK PHENOL CALIBRATION**

6.0 MODEL VERIFICATION

6.1 Verification Inputs

Following the calibration of a model, it should be tested against another set of quality data for the system to establish its predictive capabilities without using calibration principles. Alberta Environment collected a second set of data over the period of September 14 to 16 for this purpose.

For the verification run, flow data collected during the second survey were substituted into the model. This included new, lower values for incremental inflows calculated on the basis of the new flows. The concentrations of the various constituents of the incremental inflow were not changed. The values existing in the calibrated model were themselves assumed. During Survey 2 incremental inflow quality data was not collected, therefore, no firm basis existed for modifying the parameter concentrations in the incremental inflow. The headwater concentrations used for the verification run were those observed during Survey 2. Decay rates for the various non-conservative parameters were the same as those resulting from the calibration.

The flows observed in the mainstem during the second survey were 20 to 30 percent of those observed during the first survey (Table 6.1). Muriel Creek was not flowing and Marie Creek exhibited very reduced flows. This is indicative of a low flow period during which the major inflow component to the mainstem would be groundwater.

TABLE 6.1

BEAVER RIVER FLOWS, SURVEYS 1 AND 2

Station	Survey 1 Flow (m^3/s)	Survey 2 Flow (m^3/s)
Beaver River at Goodridge	1.44	0.25
Beaver River at Highway 881	1.26	0.45
Beaver River at Big Meadow	9.32	2.78
Beaver River Below Marie Creek	10.7	3.35
Beaver River at Cold Lake Reserve	12.6	3.54
Sand River at Truman	8.67	2.67

6.2 Verification Results

Graphical representations of the results of the verification analysis for the Beaver River and Marie Creek are presented in Appendix C. Those for the Sand River and Muriel Creek are not presented. At the time of the second survey there was no flow in Muriel Creek so verification would be meaningless. Since the Sand River calibration was to only two data points near the mouth, verification was not carried out.

The graphs in Appendix C illustrate a good prediction for dissolved oxygen at the low flow level in the Beaver River and in Marie Creek. The observed Marie Creek data shows the influence of plant photosynthesis and respiration which was noted earlier and is not predicted by the model in its steady state configuration.

The calibrated model over-predicts the coliform concentration observed in the mainstem during the second survey by about 20/100 ml just below the Sand. The major portion of this over-prediction in the mainstem was due to the input from the Sand River. Survey 2 data indicated that the level in the Sand for total coliform was >18/100 ml, while the calibrated model predicted approximately 50/100 ml. Reducing the headwater inputs at Caribou Lake improved the prediction slightly.

The major contributor of coliforms to the Beaver River system is surface runoff. Since the second survey was undertaken during a dry period there would have been very little surface runoff. Therefore, the coliform input to the stream would have been reduced. The higher levels noted above the confluence with the Sand are probably due to inputs from direct stock watering.

BOD values observed during Survey 2 in the mainstem varied from 8 to 9 mg/l. These are considerably higher than those observed during Survey 1. Despite these high values, however, no detrimental affect on instream dissolved oxygen concentration was observed. This is due to the high reaeration character of the stream and possibly a relatively low BOD decay rate. The high BOD is thought to be due to the winter die-back of aquatic plants which would contribute carbonaceous particulates to the stream.

Ammonia nitrogen levels were higher during Survey 2 than during Survey 1, exhibiting a range of from 0.01 to 0.09 mg/l. Despite this wide range most of the data was relatively constant along the mainstem at approximately 0.04 mg/l. This is indicative of a balanced ecosystem in which inputs equal decay and outputs. The higher levels are attributed to the decomposition of aquatic plants in localized low oxygen environments. The calibrated model under-predicted the Survey 2 results. This was probably due to an increased benthic input to the stream which was not observed during Survey 1, and therefore not calibrated.

Dissolved ortho-phosphate values were well predicted above the Sand, but over-predicted below the Sand. Either the ortho-phosphate concentration in the groundwater input decreased during the second survey or instream biological uptake had increased significantly over that observed during the first survey. The more normal values above the Sand are probably due to livestock contributions.

The results of the conservative constituents, Na and Cl, show an under-prediction of the Survey 2 results by the calibrated model. This may be due to higher incremental inflow constituent concentrations than those prevalent during the first survey.

The incremental inflow to the stream is a combination of a groundwater contribution and surface runoff. Typically, since groundwater has the greater residence time in contact with the soil, it absorbs larger amounts of these soluble conservatives than does the surface runoff. During times of medium to high flows the runoff component by virtue of its larger volume effectively dilutes the groundwater concentration resulting in a lower instream concentration. During dry periods, such as that which prevailed during Survey 2, the runoff component is reduced and the groundwater component predominates. Therefore the values observed during Survey 2 are indicative of the effects of groundwater contaminant concentrations.

Sulphate concentrations are under-predicted by the model upstream of the Sand and over-predicted downstream. This may be the result of the variability in sulphate data noted in Section 5.8.

The results of the calibrated model predictions on Marie Creek showed good agreement except for the reach downstream of the sewage treatment plant input. This was probably due to an effluent quality different from that used during calibration.

The discrepancies between calibrated model predictions and low flow observations can be explained. However, they illustrate that caution must be exercised in interpreting the results of production runs. These variations may be considered to be the degree of accuracy of the model and the results may be interpreted in this light.

The discrepancies are also representative of the difficulties in calibrating and verifying a mathematical model on a good quality stream. Analytical measurements are often near the technological limits and the easily quantifiable impacts of point sources are absent.

7.0 FUTURE WATER QUALITY

7.1 Development Scenario

The future water quality was predicted on the basis of an industrial development scenario to the year 2000, prepared by Alberta Environment. This development comprised the construction of two large heavy oil upgrading plants and ten medium sized plants (Table 7.1). Other development scenarios will be investigated at a later date.

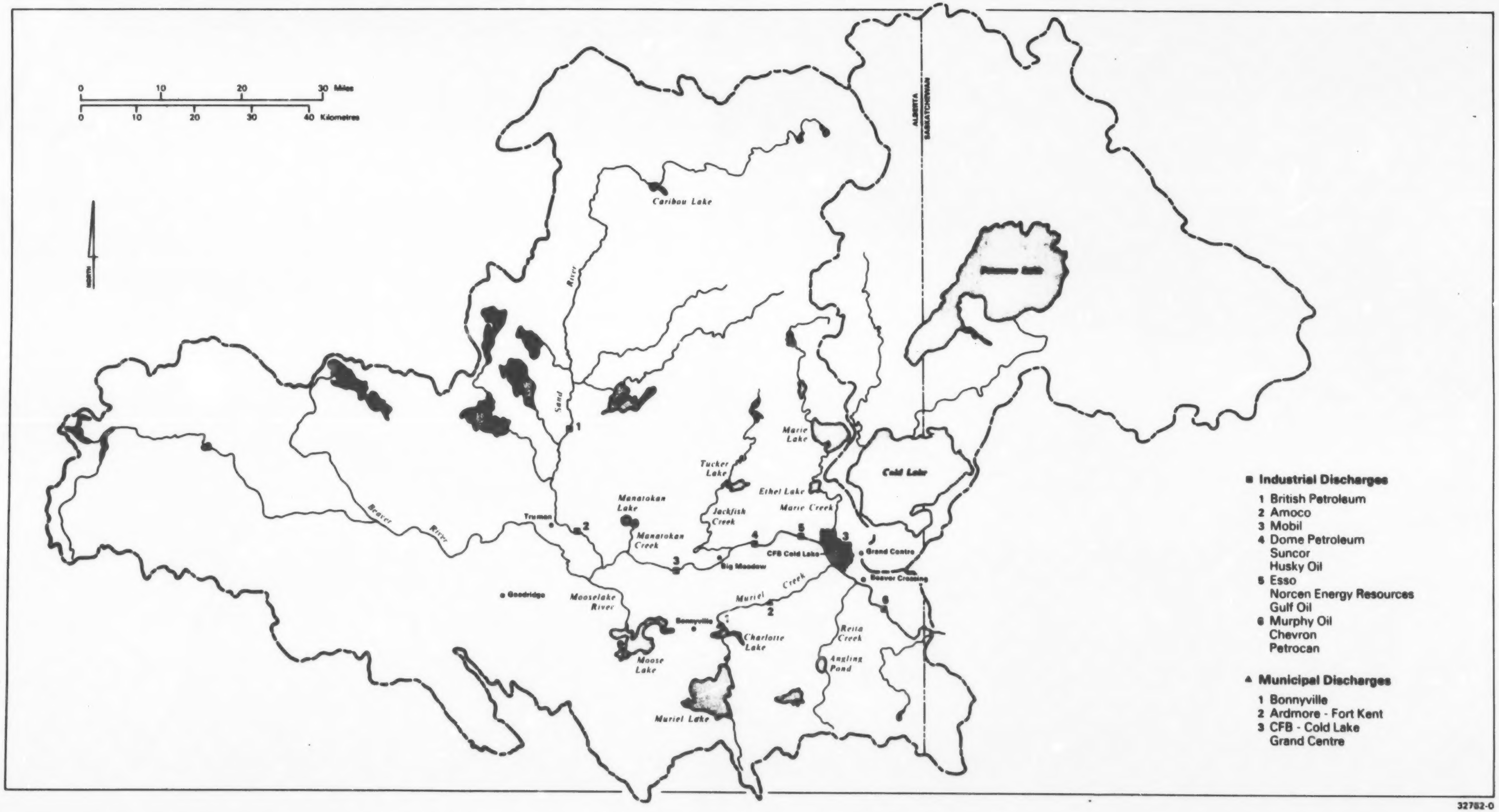
The two different size categories of upgrading plants are expected to utilize different production processes. Because of the different processes and sizes, wastewaters of different quantities and characteristics will be produced. These wastes will be discharged to the Sand and Beaver Rivers at locations shown in Figure 7-1. Table 7.1 illustrates the quantity of wastewater expected from each plant and Table 7.2 gives the estimated characteristics of the wastewater.

This level of industrial development is expected to result in an increase in the population of the basin of almost 2.5 times, to approximately 65 000. The new population will be spread among three large centres, several smaller centres and the rural area (Table 7.3).

Sanitary wastewaters generated by the increased population will be treated prior to discharge to receiving waters in the area. The predominant treatment method in this area of Alberta is by seasonal retention lagoons. Wastewater is stored in these constructed lagoons, where it is treated, by natural biological processes, for approximately six months.

Figure 7-1

Projected Waste Discharges 2000 AD


0 10 20 30 Miles
0 10 20 30 40 Kilometres
NORTH
ALBERTA
SASKATCHEWAN
Caribou Lake
Sand River
Beaver River
Truman
Goodridge
Manatokan Lake
Manatokan Creek
Tucker Lake
Jackfish Creek
Marie Lake
Ethel Lake
Marie Creek
Cold Lake
CFB Cold Lake
Big Meadow
Grand Centre
Beaver Crossing
Mooselake River
Bonnyville
Muriel Creek
Charlotte Lake
Moose Lake
Muriel Lake
Reita Creek
Angling Pond
1
2
3
4
5
6
■ Industrial Discharges
1 British Petroleum
2 Amoco
3 Mobil
4 Dome Petroleum
Suncor
Husky Oil
5 Esso
Norcen Energy Resources
Gulf Oil
6 Murphy Oil
Chevron
Petrocan
▲ Municipal Discharges
1 Bonnyville
2 Ardmore - Fort Kent
3 CFB - Cold Lake
Grand Centre

32782-0

TABLE 7.1

HEAVY OIL UPGRADING PLANTS
(Alberta Environment, March 1982)

Proponent	Wastewater Volume (m^3/d)	Receiving Water Body	Distance Upstream (km)
LARGE PLANTS			
Esso	4200	Beaver River	43
Dome	4200	Beaver River	57
MEDIUM PLANTS			
British Petroleum	721	Sand River	69
Amoco	381	Sand River	12
Mobil	359	Beaver River	80
Husky	200	Beaver River	57
Gulf	167	Beaver River	43
Chevron	181	Beaver River	18
Murphy	247	Beaver River	18
Norcen	285	Beaver River	43
Petro Canada	195	Beaver River	18
Suncor	595	Beaver River	57

TABLE 7.2

SIMULATED INDUSTRIAL WASTE DISCHARGE CONCENTRATIONS
(Alberta Environment, March 1982)

Parameter*	Large Plants	Medium Plants
Biological Oxygen Demand	0	10
Chemical Oxygen Demand	28	-
Chromium (+6)	0.01	-
Total Phosphorus (As PO_4)	0.25	-
Zinc	0.02	-
Total Suspended Solids	18.3	25
Oil and Grease	1.3	-
Sulphides	0.05	-
Ammonia - Nitrogen	3.05	-
Sulphate	2860	150
Sodium	1545	1000
Chlorides	637	1000
Phenols	0.014	-
Total Copper	0.002	-
Total Dissolved Solids	6246	4350
Silica	-	16
Magnesium	-	120
Calcium	-	180

* All concentrations in mg/L

TABLE 7.3

PROJECTED BEAVER RIVER BASIN POPULATION, 2000 AD
(Alberta Environment, March 1982)

Location	Population
LARGE CENTRES	
Bonnyville	11 680
Cold Lake - Grand Centre	31 750
CFB Cold Lake	6 500
SMALL CENTRES	
Ardmore - Fort Kent	1 390
Glendon	1 540
Mallaig	720
RURAL	11 195
	64 775

The treated wastewater is discharged to the receiving water at periods of high stream flow, usually in the spring and fall. During these periods sufficient dilution water is available to reduce the instream concentration of contaminants to acceptable levels.

The rate of discharge from these lagoons for prediction purposes was calculated as follows:

$$Q_d = \frac{q_c P\ D}{d}$$

Where; Q_d = daily flow during the discharge period, m^3/d.

q_c = daily per capita wastewater contribution, m^3/d.

P = Contributing population.

D = Storage period, days.

d = Discharge period, days.

For large centres the per capita contribution was estimated to be 0.45 m^3/d, while that from small centres was 0.29 m^3/d.

The quality of the discharges was predicted by Alberta Environment from past experience in the area (Table 7.4). Phosphorous discharges were reported as "P". However, since the model operates on dissolved ortho-phosphate 90% of the phosphorous was assumed to be ortho-phosphate. The only mechanical treatment plant in the basin services CFB - Cold Lake and discharges into Marie Creek. The location of the major municipal waste discharges in the area are shown in Figure 7-1.

7.2 Future Water Quality Prediction

Prediction of future water quality was carried out by superimposing the wastes emmanating from industrial and urban developments onto the calibrated model. The runs were carried out under low flow conditions in the basin calculated by Nanuk Engineering and Development Ltd. based upon a 7 day, 10 year low flow of the Alberta-Saskatchewan border (Table 7.5). Incremental inflow rates for the production runs were recalculated using these flows.

A total of six production runs were carried out (Table 7.6). Four were specified runs in which the low flow conditions provided were applied to summer and autumn periods with varying conditions of municipal wastewater discharge. A fifth and sixth run were carried out to test the effect of a high BOD discharge from the two large heavy oil plants on the mainstem water quality and the sensitivity of predicted high un-ionized ammonia levels in Muriel Creek to higher nitrification rates.

TABLE 7.4

SIMULATED MUNICIPAL WASTE DISCHARGE CONCENTRATIONS
(Alberta Environment, March 1982)

Parameter	Concentration (mg/L)	
BOD	25	
Suspended Solids	25	
Total Phosphorus (as P)	5	
Ortho-Phosphate (as P)	4.5	
Kjeldahl (N)	25	
Ammonia (N)	20	
TDS	217[1]	330[2]
Threshold Odour No.	4	
Sodium	40[3]	49[4]
Sulphate	65[5]	78[6]
Oil and Grease	20	
Phenol	.05	

[1] 10% added to background summer river concentration

[2] 10% added to background winter river concentration

[3] 25 mg/L added to background summer concentration

[4] 25 mg/L added to background winter concentration

[5] 50 mg/L added to backgrouud summer concentration

[6] 50 mg/L added to background winter concentration

TABLE 7.5

PREDICTED LOW FLOW CONDITIONS

Location	Flow (m^3/s)
Caribou Lake	0.52
Sand River at Truman	1.05
Sand River at Mouth	1.26
Beaver River at Goodridge	0.37
Beaver River at Mooselake River	0.47
Mooselake River at Moose Lake	0.00
Mooselake River at Mouth	0.04
Beaver River at the Sand	0.51
Manatokan Lake	0.00
Beaver River at Jackfish	1.91
Tucker Lake Outlet	0.00
Jackfish Creek at Mouth	0.03
Beaver River at Marie Creek	2.01
Ethel Lake Outlet	0.00
Marie Creek at Mouth	0.07
Beaver River at Muriel Creek	2.17
Charlotte Lake Outlet	0.04
Muriel Creek at Mouth	0.13
Beaver River at Reita Creek	2.24
Angling Lake Outlet	0.00
Reita Creek at Mouth	0.02

TABLE 7.6

PRODUCTION RUN CONDITIONS

Run No.	Season	Temperature (°C)	Municipal Lagoon Discharge	Bonnyville Waste Discharge	Nitrification Rate (Day 1)	Large Plant BOD (mg/L)
1	Summer	20 - 22	Continous	Included	0.1	0
2	Autumn	5	30 Days	Included	0.1	0
3	Summer	20 - 22	Continous	Excluded	0.1	0
4	Autumn	5	30 Days	Excluded	0.1	0
5	Summer	20 - 22	Continuous	Included	0.5	28
6	Summer	20 - 22	Continuous	Excluded	0.5	28

7.3 Predicted Water Quality, 2000 A.D.

The results of the first four production runs are summarized with respect to compliance with the applicable water quality objectives in Tables 7.7 to 7.10. Both the Surface Water Quality Objectives of the Province of Alberta (1977) and the interim Prairie Provinces Water Board requirements are shown. Graphical results for Run No. 1 are shown in Appendix D. Those of the other runs are not presented in this report since they are all very similar. (These results are submitted under separate cover.)

The results of all six runs indicated that the Sand River will not be appreciably affected by the two industrial discharges located on its lower reaches. Similarly, the upper Beaver River will be unaffected.

The concentrations of chloride, sodium and sulphate on the Beaver River between the Sand River and Marie Creek were predicted to exceed the objectives. There will be, in this stretch, an industrial discharge which will include a large oil upgrading plant. It is the contribution from this discharge which resulted in the exceedence of the criteria. Since these are conservative constituents the high concentrations were carried downstream and added to by the two industrial discharges farther downstream. The predicted levels of chloride, sodium and sulphate at the Saskatchewan border varied from 30 to 46 mg/l, 79 to 100 mg/l and 121 to 150 mg/l, respectively. The sodium and sulphate values exceeded the PPWB objectives by factors of greater than two. Chloride levels were approximately equal to the PPWB requirements at the Saskatchewan border in two runs. However, within the limits of accuracy of the model these should be considered to exceed the criteria.

TABLE 7.7

EXCEEDENCE OF WATER QUALITY OBJECTIVES RUN NO. 1, SUMMER LOW FLOW

Location	Dissolved Oxygen		Phosphorus		Un-ionized Ammonia	Chloride	Sodium	Sulphate	Phenol
	Alta (5 mg/L)	PPWB (6 mg/L)	Alta (0.15 mg/L PO_4)	PPWB (0.22 mg/L)	PPWB (0.02 mg/L)	PPWB (30 mg/L)	PPWB (40 mg/L)	PPWB (50 mg/L)	PPWB (0.005 mg/L)
Beaver River									
1. Above Sand									
2. Between Sand and Marie						X	X	X	
3. Below Marie			X	X	X	X	X	X	X
Sand River									
Marie River			X	X	X	X	X	X	X
Muriel Creek	X	X	X	X	X	X	X	X	X

TABLE 7.8

EXCEEDENCE OF WATER QUALITY OBJECTIVES RUN NO. 2, AUTUMN LOW FLOW

Location	Dissolved Oxygen		Phosphorus		Un-ionized Ammonia	Chloride	Sodium	Sulphate	Phenol
	Alta (5 mg/L)	PPWB (6 mg/L)	Alta (0.15 mg/L PO_4)	PPWB (0.22 mg/L)	PPWB (0.02 mg/L)	PPWB (30 mg/L)	PPWB (40 mg/L)	PPWB (50 mg/L)	PPWB (0.005 mg/L)
Beaver River									
1. Above Sand									
2. Between Sand and Marie						X	X	X	
3. Below Marie			X	X	X	X	X	X	X
Sand River									
Marie River			X	X	X		X	X	X
Muriel Creek	X	X	X	X	X	X	X	X	X

TABLE 7.9

EXCEEDENCE OF WATER QUALITY OBJECTIVES RUN NO. 3, SUMMER LOW FLOW, EXCLUDING BONNYVILLE

Location	Dissolved Oxygen		Phosphorus		Un-ionized Ammonia	Chloride	Sodium	Sulphate	Phenol
	Alta (5 mg/L)	PPWB (6 mg/L)	Alta (0.15 mg/L PO_4)	PPWB (0.22 mg/L)	PPWB (0.02 mg/L)	PPWB (30 mg/L)	PPWB (40 mg/L)	PPWB (50 mg/L)	PPWB (0.005 mg/L)
Beaver River									
1. Above Sand									
2. Between Sand and Marie						X	X	X	
3. Below Marie			X	X	X	X	X	X	X
Sand River									
Marie River			X	X	X		X	X	X
Muriel Creek		X	X	X	X	X	X	X	X

TABLE 7.10

EXCEEDENCE OF WATER QUALITY OBJECTIVES RUN NO. 4, AUTUMN LOW FLOW, EXCLUDING BONNYVILLE

Location	Dissolved Oxygen		Phosphorus		Un-ionized Ammonia	Chloride	Sodium	Sulphate	Phenol
	Alta (5 mg/L)	PPWB (6 mg/L)	Alta (0.15 mg/L PO_4)	PPWB (0.22 mg/L)	PPWB (0.02 mg/L)	PPWB (30 mg/L)	PPWB (40 mg/L)	PPWB (50 mg/L)	PPWB (0.005 mg/L)
Beaver River									
1. Above Sand									
2. Between Sand and Marie						X	X	X	
3. Below Marie			X	X	X	X	X	X	X
Sand River									
Marie River			X	X	X	X	X	X	X
Muriel Creek			X	X	X	X	X	X	X

Below Marie and Muriel Creeks, the Beaver River showed the effects of municipal discharges to these tributaries in high phosphorus and un-ionized ammonia levels. Over the first four runs, dissolved ortho-phosphate levels varied from 0.49 to 1.6 mg/l. These concentrations were in excess of the limit of 0.15 mg/l-PO_4 set by Alberta Environment, and the maximum allowable level of 0.22 mg/l-P set by the PPWB.

The percentage of the ammonia present which is un-ionized is directly proportional to both pH and temperature and is of great concern due to its high toxicity. QUAL-II simulates the behaviour of total ammonia only and hence the un-ionized fraction must be calculated.

Naquadat data indicated that the pH of the lower Beaver River varies from 7.0 to 8.5. Using these pH limits and the ranges of ammonia concentrations predicted for Marie and Muriel Creeks and the lower Beaver River, the un-ionized ammonia concentrations were calculated (Table 7.11). Concentrations in excess of the PPWB limit of 0.02 mg/l were predicted under all simulation conditions at pH = 8.5.

At pH = 7, the un-ionized ammonia criteria were not exceeded in the lower Beaver River. However, the majority of the Naquadat pH measurements between 1966 and 1980 were above 7.5. Therefore, un-ionized ammonia limits can be expected to be exceeded frequently under this development scenario.

The major contributors of ammonia are the treated municipal discharges. Since these discharges travel through relatively small, slow moving streams significant numbers of nitrifiers might be expected to be present in the stream. Accordingly, Runs 5 and 6 were run at a high nitrification rate. The

TABLE 7.11

UN-IONIZED AMMONIA CONCENTRATIONS

Simulation Run No.		Marie Creek Total NH_3 Range (mg/L)	Marie Creek Un-ionized NH_3 Range pH = 7.0 (mg/L)	Marie Creek pH = 8.5 (mg/L)	Muriel Creek Total NH_3 Range (mg/L)	Muriel Creek Un-ionized NH_3 Range pH = 7.0 (mg/L)	Muriel Creek pH = 8.5 (mg/L)	Lower Beaver River Total NH_3 Range (mg/L)	Lower Beaver River Un-ionized NH_3 Range pH -7.0 (mg/L)	Lower Beaver River pH = 8.5 (mg/L)
1	High	15.24	(0.06)	(1.68)	11.98	(0.05)	(1.32)	1.78	0.007	(0.196)
	Low	12.49	(0.05)	(1.37)	5.81	(0.02)	(0.64)	1.34	0.005	(0.147)
22°C										
2	High	19.02	(0.025)	(0.72)	18.0	(0.023)	(0.68)	7.82	0.01	(0.30)
	Low	18.15	(0.024)	(0.69)	14.38	0.019	(0.55)	6.96	0.009	(0.26)
5°C										
3	High	15.24	(0.06)	(1.68)	1.24	0.005	(0.136)	1.53	0.006	(0.17)
	Low	12.49	(0.05)	(1.37)	0.73	0.003	(0.08)	1.07	0.004	(0.18)
22°C										
4	High	19.02	(0.025)	(0.72)	5.65	0.007	(0.21)	6.95	0.009	(0.26)
	Low	18.15	(0.024)	(0.69)	3.81	0.005	(0.14)	5.91	0.008	(0.22)
5°C										
5	High	15.24	(0.06)	(1.68)	11.78	(0.05)	(1.3)	1.56	0.006	(0.17)
	Low	12.5	(0.05)	(1.37)	2.72	0.01	(0.30)	1.17	0.005	(0.13)
22°C										
6	High	15.24	(0.06)	(1.68)	1.21	0.005	(0.13)	1.53	0.006	(0.17)
	Low	12.5	(0.05)	(1.37)	.47	0.002	(0.05)	1.06	0.004	(0.12)
22°C										

* Parenthesis indicate exceedence at PPWB limits.

table shows that toxic levels may still be exceeded in Muriel and Marie Creeks as well as the Beaver River at the higher pH.

Because of the shallow, fast flowing nature of the Sand and Beaver Rivers, the reaeration rates (K_2) are sufficiently high to maintain dissolved oxygen levels well above the Alberta and PPWB criteria throughout their lengths. The diurnal affect of photosynthesis in the stream may be expected to decrease the near saturation values observed, by about 1 to 1.5 mg/l (Figure 5-1).

Dissolved oxygen on Muriel Creek, dipped below both criteria in Runs 1 and 2, due to the discharge of municipal waste from Bonnyville and Ardmore-Fort Kent. If Bonnyville waste were not discharged to Muriel Creek, only the higher PPWB criterion of 6 mg/l would be violated, and then only in the summer (Runs 3 and 4). The diurnal photosynthesis cycle may, however, result in the reduction of dissolved oxygen to below the Alberta criterion in the summer.

On Marie Creek, no violation of dissolved oxygen standards are predicted. However, it is known that this stream, below the municipal outfalls is slow, and may become highly productive due to the nutrients discharged with the sewage. Therefore, photosynthetic activity may result in critically low dissolved oxygen levels in the summer period.

Phenol concentrations in Marie and Muriel Creeks and the lower Beaver River exceeded the criteria in each run. This was due to the high phenol concentrations in the municipal discharges to the two tributary creeks.

7.4 WINTER WATER QUALITY PREDICTION

7.4.1 General

Due to its northern location, The Beaver River Valley often freezes over completely in the winter. During this time, overland runoff is minimal, and very small contributions of flow are available. Accordingly, the lowest flow experienced in the river is during winter. Because of this low flow and the accompanying low dilution capacity, discharges of industrial and municipal waste in the future, may result in critical conditions with respect to particular constituents.

The national water quality data system, NAQUADAT, maintains one sample collection point on the Beaver River, at Beaver Crossing. This location was also a sampling station on the surveys carried out in conjunction with the summary calibration of Qual-II. The NAQUADAT data at Beaver Crossing has a short period of record, however it is known that the dissolved oxygen during the winter, diminishes to less than 2 mg/L from the high saturated to supersaturated value prevalent in the summer. Nitrate and nitrite increase in the system, possibly due to the slow degradation of ammonia or a high concentration of these species in the ground water which forms the only incremental input to the river during the winter. The latter is more likely, since at the temperatures existing in the river during the winter, nitrification would be virtually zero. Changes in concentration between summer and winter of other constituents sampled at Beaver Crossing are probably due to the lack of surface runoff which would either dilute or increase the levels of various constituents. In this matter, the concentration of conservative materials in the river during the winter may be considered to be a composite sample of the ground water quality. It may also be possible to extrapolate this winter value of ground water quality to reflect the summer quality of incremental inflows.

Given the low flow and the potential for continuous waste discharge in the period of future industrialization, it is important to predict the effect of these winter discharges. However, since no survey has been carried out during the winter because of the excessive cost and the logistical difficulties, a rigorous calibration of the Qual-II model for winter conditions, is not possible. Little work has been carried out in simulating the winter conditions in rivers which freeze-over. However, it is felt that with judicious assumptions based on well known phenomena, the predictions made by an uncalibrated model can provide a conservative basis upon which to make regulatory decisions. Confidence limits around these predictions based on the winter simulation, must be acknowledged as being very wide. For more rigorous regulatory control, a detailed winter calibration would be required.

7.4.2 Winter Model Set-up

The assumptions employed in setting up the model for winter simulation, are mainly process oriented. They are based on the fact that the river freezes over completely and account for the physical effect of the ice with respect to reaeration and waterflow, and the effect of the low temperatures upon biological processes.

Winter flow conditions received from Nanuk Engineering represent 1 and 10 year low flow phenomena during the winter, and indicate that flow and complete ice cover will have a large effect on the friction factor and the channel length of the river. At very low flow, the water finds its own route around minor topographical bottom features and ice blocks, effectively lengthening the channel and the travel time in the river and increasing the friction.

Freezing will also produce additional friction due to the surface ice cover. Chow (1959) described an empirical method whereby the flow in an ice covered channel can be corrected, to account for the ice conditions. By rearranging the equations used in this method, the friction factor for the various reaches of the Beaver River was corrected for the ice conditions and the low flow.

Biological processes which exist in the river during warm weather conditions are temperature dependent. Accordingly, the reaction rates used in the equations for the Qual-II model must be modified to reflect the reduced rates. In certain cases, it may be assumed that these reaction rates are effectively zero.

Zero reaction rate assumptions were applied to the winter simulation of algae, the nitrogen cycle and the phosphorous cycle. The increase in nitrates over the winter at Beaver Crossing is attributed to the concentration of nitrates in the ground water. This reflects the belief held by a large number of researchers that the nitrification of ammonia is highly temperature dependent and decreases to virtually zero below 5°C. The phosphorous cycle can be considered to be in-active in the winter, since the only sink for phosphorous in the Qual-II model, is algae. K1 may be corrected using the Streeter-Phelps correction equation for small changes in temperature. However Zanoni (1967) has indicated that this correction method is not valid for temperature approaching 0°C. The BOD winter decay rate was corrected to the zero degree range by using a method similar to that shown by McBean (1979).

McBean corrected observed decay rates at 20°C to 0.4°C by Streeter-Phelps and compared them with those measured at 0.4°C. He found that the measured and estimated values were vastly different and illustrated the ratio of these in his paper. The values used in the summer calibration of Qual-II on the Beaver River system were corrected by Streeter-Phelps to 0.4°C. Then, the ratio of K1 estimated to K1 measured, found by McBean, was used to further correct the estimated values for input into the winter simulation.

Since complete ice cover exists on the Beaver River during the winter, the reaeration rate, K2, was set to zero. The large reduction in dissolved oxygen in the Beaver River during the winter may result from the lack of reaeration or the slow degradation of sediment BOD within the system at these low temperatures. Accordingly, for the winter simulation benthic BOD was not set at 0, rather, a value was selected from Tetra Tech (1978) and corrected to 0°C by the McBean method.

Input to the calibrated Qual-II model for the Beaver River was modified according to the above assumptions. The new model was run on the University of Alberta computer system using a remote terminal from the Alberta Environment offices in Edmonton. At the time of writing, the detailed results of these runs were not available, however, they will be available and presented by Alberta Environment following completion of the winter simulation runs. It is proposed to use the various development scenarios and the resulting municipal and industrial effluents from these scenarios to predict winter conditions on the Beaver River. Caution must be used in interpreting these values since a large amount of input data is based on assumptions rather than hard calibration data.

8.0 CONCLUSIONS

1) The existing water quality in the Beaver and Sand Rivers is good and typical of wilderness rivers relatively unimpacted by anthropogenic influences.

2) The existing water quality of Marie and Muriel Creeks is affected by municipal waste discharges and contaminant inputs from agricultural landuse. Their overall quality is of a lower level than the two main rivers in the basin, with Muriel Creek experiencing the greatest impact.

3) In the absence of point source discharges of non-conservative substances the values of decay coefficients for the various water quality parameters are difficult to identify exactly. However, within these limitations, the model may be said to describe the behaviour of the instream chemical constituents of the Beaver River Basin.

4) The projected industrial development scenario to the year 2000 AD will not result in an exceedence of water quality criteria set by Alberta Environment or the Prairie Provinces Water Board (PPWB) in the Sand River or above the Sand on the Beaver River.

5) Between the Sand River and Marie Creek the industrial discharges will add sufficient chloride, sulphate and sodium to exceed the Alberta Environment and PPWB criteria for these substances.

6) Alberta Environment and PPWB criteria for phosphorus, un-ionized ammonia, chloride, sodium, sulphate and phenol will be exceeded on the Beaver River below Marie Creek under low flow conditions. This will result primarily from the municipal waste discharges to Marie and Muriel Creeks which cause high levels of phosphorus, un-ionized ammonia and phenol in the mainstem.

7) The water quality of the lower portion of Marie Creek will exceed the criteria for phosphorus, un-ionized ammonia, sodium, sulphate and phenol due to continuous and intermittent discharge of municipal waste from Cold Lake-Grand Centre and the Canadian Forces Base. (Current plans call for the relocation of the Cold Lake-Grand Centre discharge to the Beaver River).

8) Continuous and intermittent discharges of treated municipal waste from Bonnyville and Ardmore-Fort Kent to Muriel Creek, will result in exceedences of all Alberta Environment and PPWB water quality criteria in the stream at low flow conditions.

9) The dissolved oxygen levels in Muriel Creek are expected to be below 5 mg/l, due to the discharge of municipal wastes.

10) The removal of the Bonnyville waste discharge from Muriel Creek will improve the quality of the upper reaches with respect to phosphorus, un-ionized ammonia, dissolved oxygen and phenol. However, the Ardmore-Fort Kent discharge results in high levels of these parameters in the lower reaches.

11) An increased nitrification rate in Marie and Muriel Creeks would result in lower instream un-ionized ammonia concentrations. However, the un-ionized ammonia criteria of 0.02 mg/l may still be exceeded under certain conditions of pH and temperature, in both the creeks and the lower Beaver River.

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APPENDIX A

SAMPLING SURVEY RESULTS

TABLE A-1

Reconnaissance Field Trip Sampling Data

Station	DO	BOD_5	NO_2-N	NO_3-N	TKN	Total	Total P	Ortho P	Phenol	Cl	SO_4	Na
Reita Creek Near Mouth	8.9	1.5	0.011	0.022	1.22	0.070	0.220	0.102	0.005	6.8	38.8	31.0
Beaver @ Beaver Crossing	9.5	2.2	0.003	0.003	0.82	0.020	0.080	0.022	0.002	3.7	14.8	17.4
Mooselake River @ Mouth	15.1	3.2	0.079	2.12	1.43	0.050	0.930	0.86	0.001	11.0	19.5	19.9
Beaver, above Marie Creek	13.8	1.6	0.003	0.005	0.83	0.020	0.083	0.017	0.003	3.3	14.2	16.1
Muriel Creek @ East of Ardmore	7.1	3.5	0.003	0.003	2.93	0.060	0.380	0.350	0.002	38.0	140	110.0
Beaver, above Jackfish Creek	10.2	3.1	0.003	0.003	0.72	0.010	0.075	0.016	0.009	2.6	13.5	13.7
Jackfish Creek		2.5	0.003	0.003	1.4	0.010	0.093	0.016	0.003	1.5	12.3	20.9
Manatokan Creek Near Mouth	9.2	1.8	0.023	0.149	6.1	0.130	0.113	0.017	0.005	3.7	47.0	23.8
Beaver, above Manatokan	10.2	3.0	0.003	0.003	0.69	0.002	0.043	0.013	.002	2.0	12.2	12.6
Sand River @ Truman	9.4	1.6	0.003	0.003	0.71	0.030	0.048	0.004	-	1.0	4.5	9.4
Beaver, South of Truman	11.0	1.5	0.008	0.004	1.26	0.050	0.138	0.048	-	5.0	29.2	22.5
Mooselake River	9.7	3.8	0.009	0.013	1.94	0.120	0.085	0.034	-	9.3	64.0	42.0

TABLE A-2

Station Number	Beaver @ Goodridge Bridge	Beaver @ Hwy 881 Bridge	Beaver @ South of Truman	Beaver Downstream of Truman Bridge	Beaver Upstream of Mooselake River	Sand River @ Truman	Mooselake River @ Mooselake	Mooselake River @ Mouth	Beaver Upstream of Sand Confluence	Sand River @ Mouth	Beaver River Below Sand Confluence
River (km)	143	135	124	116	107	18	14	0	101.5	0	101
BOD (mg/L)	1.0	1.0	1.0	1.0	1.0	1.0	4.6	1.0	-	1.0	1.0
SO_4 (mg/L)	8.0	12.0	17.0	17.0	21.0	5.0	69.0	38.0	-	<5	<5
Phenol (mg/L)	0.002	0.003	0.002	0.003	0.002	0.002	0.003	0.003	-	0.003	0.002
Na (mg/L)	15.0	15.0	20.0	19.0	21.0	6.0	53.0	30.0	-	6.0	10.0
Dissolved Ortho-phosphate as P (mg/L)	0.125	0.120	0.12	0.115	0.104	0.016	0.01	0.068	-	0.07	0.033
Total phosphorus (mg/L)	0.29	0.25	0.23	0.33	0.25	0.11	0.16	0.20	-	0.11	0.12
NH_3 (mg/L)	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.17	-	0.20	0.02
NO_2 (mg/L)	0.003	0.004	0.004	0.051	0.002	0.002	0.002	0.016	-	0.002	0.002
NO_3 (mg/L)	0.004	0.009	0.009	0.001	0.004	-	0.003	0.095	-	0.004	0.002
Chlorophyll A (mg/L)	1.61	3.01	5.62	7.63	13.65	9.64	16.06	1.84	-	7.03	10.24
Total Coliform (No/100 ml)	164	87	65	>80	83	30	2	>80	-	52	18
Fecal Coliform (No/100 ml)	27	25	48	>80	55	15	12.8	>80	-	19	18
Chloride (mg/L)	3.0	3.0	4.0	3.0	4.0	1.0	12.0	8.0	-	1.0	2.0
Total Kjeldahl Nitrogen (mg/L)	1.15	1.13	1.09	1.16	1.2	0.58	2.28	2.05	-	0.62	0.80
Dissolved O_2 (mg/L)	8.7	7.5	8.8	8.0	8.6	8.5	24.8	13.7	-	8.9	8.9
Temperature °C	20.8	20.6	21.5	19.5	21.3	21.0	-	23.2	-	21.9	21.6
Dissolved Oxygen Saturation (%)	97	83	100	87	96	94	152	157	-	101	101
Time (hrs)	1430	1500	1450	1700	1745	1630	-	1810	-	1900	1930

TABLE A-2 (Cont'd...2)

	Beaver @ Hwy 890 Bridge	Beaver River @ Big Meadow	Jackfish Creek @ Tucker Outlet Headwaters	Jackfish Creek Near Mouth	Beaver River @ Hwy 892 Bridge	Beaver Upstream Marie Creek	Beaver Downstream Marie Creek	Marie Creek @ Ethel Creek	Marie Creek @ Hwy 55	Marie Creek @ CFB-Cold Lake	Marie Creek @ Mouth
River (km)	78	686	67	0	54	40	39	27	10.3	1	0
BOD (mg/L)	1.9	2.0	5.4	2.0	2.1	2.6	2.6	1.4	1.4	2.2	2.2
SO_4 (mg/L)	12.0	13.0	<5	17	8.0	8.0	21.0	<5	8.0	12.0	25.0
Phenol (mg/L)	0.002	<0.002	-	0.002	0.002	0.002	0.002	<.002	0.003	<0.002	0.002
Na (mg/L)	11.0	11.0	20.0	23.0	11.0	13.0	11.0	16.0	14.0	24.0	24.0
Dissolved Ortho-phosphate as P (mg/L)	0.024	0.022	0.013	0.043	0.024	0.023	0.022	0.004	0.053	0.944	0.944
Total phosphorus (mg/L)	0.11	0.19	0.14	0.18	0.13	0.01	0.07	0.04	0.11	0.98	0.93
NH_3 (mg/L)	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.01	0.02	0.02
NO_2 (mg/L)	0.002	0.002	<.001	0.002	0.002	<.001	<.001	<.001	0.003	0.27	0.021
NO_3 (mg/L)	0.001	0.001	-	0.001	-	-	-	-	-	1.316	1.493
Chlorophyll A (mg/L)	9.03	8.67	88.33	1.34	2.85	12.85	10.92	3.69	1.64	17.67	10.92
Total Coliform (No/100 ml)	34	26	-	153	95	16	7	54	50	19	68
Fecal Coliform (No/100 ml)	11	11	-	102	80	11	2	6	40	18	42
Chloride (mg/L)	3.0	2.0	1.0	1.0	2.0	2.0	2.0	3.0	3.0	14.0	13.0
Total Kjeldahl Nitrogen (mg/L)	0.70	0.72	1.6	1.2	0.78	0.74	0.69	0.82	1.20	1.27	1.20
Dissolved O_2 (mg/L)	8.7	8.9		6.5	8.8	8.7	9.0	8.8	4.0	11.8	12.9
Temperature °C	23.1	23.4	24	18.8	23.9	24.1	24.4	23.2	18.6	20.0	23.7
Dissolved Oxygen Saturation (%)	100	102	-	69	104	102	106	101	43	128	151
Time (hrs)	2010	1530	1420	1620	1645	1500	1515	1055	1130	1200	1430

TABLE A-2 (Cont'd...3)

	Muriel Creek @ Charlotte Lake	Muriel Creek South of Hwy 28	Muriel Creek Hwy 28 Bridge	Muriel Creek Near Mouth	Beaver River Above Muriel Creek	Beaver River @ Hwy 28	Reita Creek @ Angling Pond	Reita Creek @ Mouth	Beaver River @ Saskatchewan Border
River (km)	38	22	13	10	37	28	33	0	0
BOD (mg/L)	13.1	-	4.0	-	-	1.6	1.8	2.4	3.2
SO_4 (mg/L)	355	-	441	426	-	17.0	28.0	34.0	12.0
Phenol (mg/L)	0.013	-	0.006	0.008	-	0.002	0.002	0.003	<.002
Na (mg/L)	221	-	293	266	-	13.0	29.0	39.0	13.0
Dissolved Ortho-phosphate as P (mg/L)	0.013	-	0.24	-	-	0.02	0.10	0.233	0.032
Total phosphorus (mg/L)	0.44	-	0.40	-	-	0.08	0.16	0.31	0.10
NH_3 (mg/L)	0.02	-	0.02	-	-	0.02	0.02	0.1	0.02
NO_2 (mg/L)	0.004	-	0.004	-	-	0.001	0.009	0.005	<.001
NO_3 (mg/L)	0.003	-	-	-	-	-	0.004	0.043	0.001
Chlorophyll A (mg/L)	2.48	0.99	0.61	-	-	10.92	2.41	-	3.98
Total Coliform (No/100 ml)	920	243	334	-	-	9	238	30	13
Fecal Coliform (No/100 ml)	350	172	252	-	-	9	159	30	13
Chloride (mg/L)	37.0	-	8.6	8.0	-	2.0	4.0	11.0	2.0
Total Kjeldahl Nitrogen (mg/L)	7.65	-	3.6	-	-	0.72	1.27	1.23	0.74
Dissolved O_2 (mg/L)									
Temperature °C	6.0	9.0	9.5	5.6	9.0	8.8	9.1	7.7	8.4
Dissolved Oxygen	19.7	-	-	21.7	-	24.6	-	24.0	24.9
Saturation (%)	65	-	-	64	-	105	-	91	100
Time (hrs)	1000	-	-	1300	-	1630	-	1700	2130

TABLE A-3

Station Number	Beaver @ Goodridge Bridge	Beaver @ Hwy 881 Bridge	Beaver @ South of Truman	Beaver Downstream of Truman Bridge	Beaver Upstream of Mooselake River	Sand River @ Truman	Mooselake River @ Mooselake	Mooselake River @ Mouth	Beaver Upstream of Sand Confluence	Sand River @ Mouth	Beaver River Below Sand Confluence
River km	143	135	124	116	107	18	14	0	101.5	0	101
BOD (mg/L)	8.7	8.6	9.3	8.5	8.1	9.7	<1.0	5.8	8.8	8.9	9.0
SO_4 (mg/L)	13.0	25.0	31.0	<5.0	35.0	15.0	104.0	48.0	33.0	23.0	10
Phenol (mg/l)	<.002	<.002	<0.002	<0.003	<0.002	<0.002	0.008	0.003	0.003	<0.002	<0.002
Na (mg/l)	22.0	25.0	41.0	13.0	42.0	14.0	73.0	44.0	42.0	31.0	17.0
Dissolved Ortho-phosphorus as P (mg/L)	0.073	0.053	0.043	0.058	0.058	0.006	0.023	0.031	0.05	0.006	0.008
Total phosphorus (mg/L)	0.140	0.13	0.12	0.130	0.120	<0.020	0.210	0.140	0.13	0.100	0.070
NH_3 (mg/L)	0.024	0.043	0.013	0.041	0.040	0.024	0.042	0.190	0.089	0.094	0.023
NO_2 (mg/L)	0.002	0.003	<.001	<0.001	<0.001	<0.001	0.002	0.021	0.003	<0.001	0.002
NO_3 (mg/L)	0.002	0.011	0.003	0.004	<0.001	0.005	0.001	0.119	0.008	0.004	0.003
Chlorophyll A (mg/L)	1.3	5.8	5.1	3.5	2.3	1.6	88.3	35.8	2.9	1.3	1.2
Total Coliform (No/100 ml)	51	28	22	57	25	9	2	>52	11	>18	>8
Fecal Coliform (No/100 ml)	42	19	12	50	24	8	1	>160	3	18	8
Chloride mg/L	5.0	5.0	8.0	2.0	8.0	2.0	15.0	11.0	9.0	6.0	2.0
Total Kjeldahl Nitrogen (mg/L)	1.01	1.0	1.16	1.11	1.0	0.74	5.3	2.6	0.94	0.90	0.77
Dissolved O_2 (mg/L)	10.8	11.0	12.8	9.2	4.2	11.8	-	15.0	8.4	9.8	11.6
Temperature °C	12.5	13.4	13.2	13.5	13.5	15.0	-	14.5	14.7	14.7	14.1
Dissolved Oxygen											
Saturation (%)	101	105	121	87	87	116	-	146	83	94	112
Time (hrs)	1330	1400	1430	1500	1530	1515	-	1615	1730	1740	1815

TABLE A-3 (Cont'd...2)

	Beaver @ Hwy 890 Bridge	Beaver River @ Big Meadow	Jackfish Creek @ Tucker Outlet Headwaters	Jackfish Creek Near Mouth	Beaver River @ Hwy 892 Bridge	Beaver Upstream Marie Creek	Beaver Downstream Marie Creek	Marie Creek @ Ethel Creek	Marie Creek @ Hwy 55	Marie Creek @ CFB-Cold Lake
River km	78	68.6	67	0	54	40	39	27	10.3	1
BOD (mg/L)	9.1/9.7	9.6	No	No	9.2/8.2	8.8	9.0	8.5	4.2	8.4
SO_4 (mg/L)	10/10	13.0	Flow	Flow	13/14	14.0	9.0	6.0	10.0	23.0
Phenol (mg/L)	<0.002	<0.003			<0.002/<0.002	<0.002	<0.002	0.005	0.007	<0.002
Na (mg/L)	19/18	20.0			20/21	22.0	22.0	17.0	–	33.0
Dissolved Ortho-phosphorus as P (mg/L)	0.007/0.007	0.009			0.016/0.009	0.007	0.009	<0.002	0.024	1.60
Total phosphorus (mg/L)	<0.02/0.04	0.030			0.03/0.03	0.030	0.020	<0.020	0.070	2.60
NH_3 (mg/L)	0.021/0.047	0.048			0.059/0.052	0.036	0.046	0.057	0.076	0.048
NO_2 (mg/L)	<0.001/0.002	0.002			0.002/0.002	0.002	0.002	0.003	0.003	0.028
NO_3 (mg/L)	0.006/0.004	0.005			0.003/0.001	0.001	0.006	–	–	3.37
Chlorophyll A (mg/L)	1.3/1.61	1.9			1.0/1.3	1.0	1.3	2.25	4.2	41.3
Total Coliform (No/100 ml)	21	10			14/48	16	20	>320	5	89
Fecal Coliform (No/100 ml)	17	6			12/20	4	9	44	5	19
Chloride mg/L	4.0/3.0	3.0			4.0/3.0	3.0	3.0	4.0	4.0	23.0
Total Kjeldahl Nitrogen (mg/L)	0.84/0.80	0.79			0.84/0.80	0.80	0.84	0.8	0.86	1.13
Dissolved O_2 (mg/L)	12.2/11.0	11.6			11.2/10.2	11.4	11.4	9.6	4.0	15.0
Temperature °C	14.5/15.0	14.0			1.40/11.0	15.0	14.5	15.0	14.0	15.0
Dissolved Oxygen Saturation (%)	113/108	112			108/92	112	111	94	38	147
Time (hrs)	1845/1240	1430			1520/1030	1445	1425	1615	1600	1520

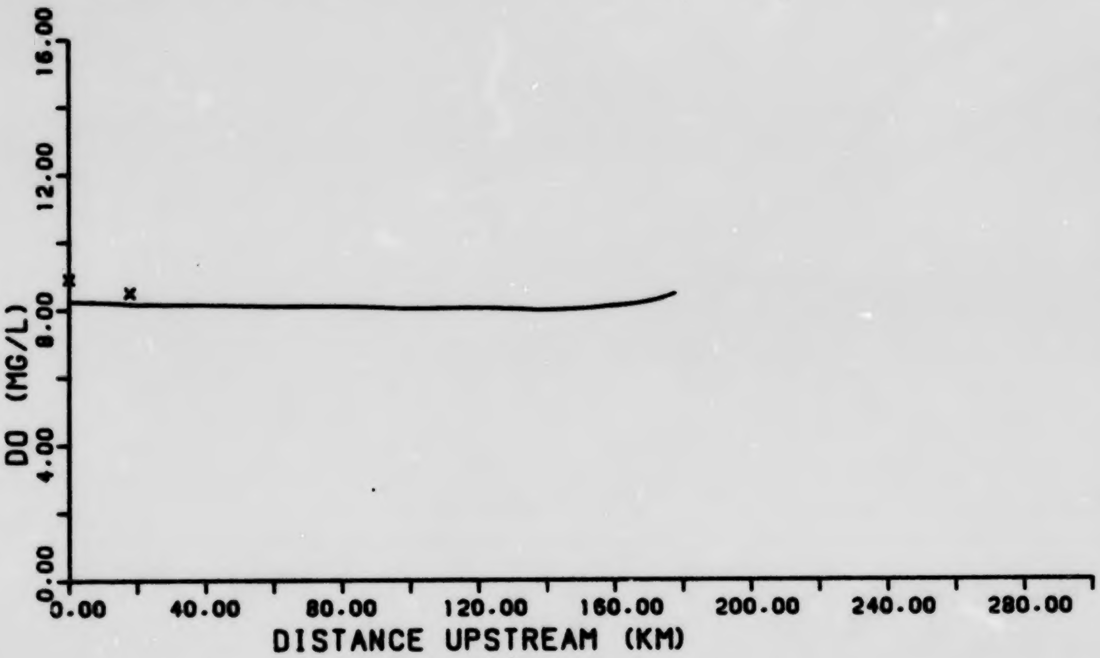
TABLE A-3 (Cont'd...3)

	Marie Creek @ Mouth	Muriel Creek @ Charlotte Lake	Muriel Creek South of Hwy 28	Muriel Creek Hwy 28 Bridge	Muriel Creek Near Mouth	Beaver River Above Muriel Creek	Beaver River @ Hwy 28	Reita Creek @ Angling Pond	Reita Creek @ Mouth	Beaver River Saskatchewan Border
River km	0	38	22	13	10	37	28	33	0	0
BOD (mg/L)	8.2	No	No	No	5.6	-	8.1	No	No	8.4
SO_4 (mg/L)	23.0	Flow	Flow	Flow	391	-	18	Flow	Flow	9.0
Phenol (mg/L)	0.003				0.012	-	<.002			<.002
Na (mg/L)	33.0				293	-	21.0			23.0
Dissolved Ortho-phosphorus as P (mg/L)	1.6				0.245	-	0.018			0.014
Total phosphorus (mg/L)	1.98				0.400	-	0.030			0.03
NH_3 (mg/L)	0.051				0.064	-	0.040			0.039
NO_2 (mg/L)	0.027				0.002	-	0.002			0.002
NO_3 (mg/L)	3.37				-	-	0.019			0.007
Chlorophyll A (mg/L)	12.5				1.3	-	1.90			1.6
Total Coliform (No/100 ml)	143				3	-	15.0			32
Fecal Coliform (No/100 ml)	24				3	-	3			5
Chloride (mg/L)	34.0				84.0	-	3.0			3.0
Total Kjeldahl Nitrogen (mg/L)	0.97				6.1	-	0.88			0.79
Dissolved O_2 (mg/L)	12				5.6	-	11.8			11.0
Temperature °C	125				10.0	-	15.0			12.0
Dissolved Oxygen Saturation (%)	112				50		116			102
Time (hrs)	1545				1100		1630			1130

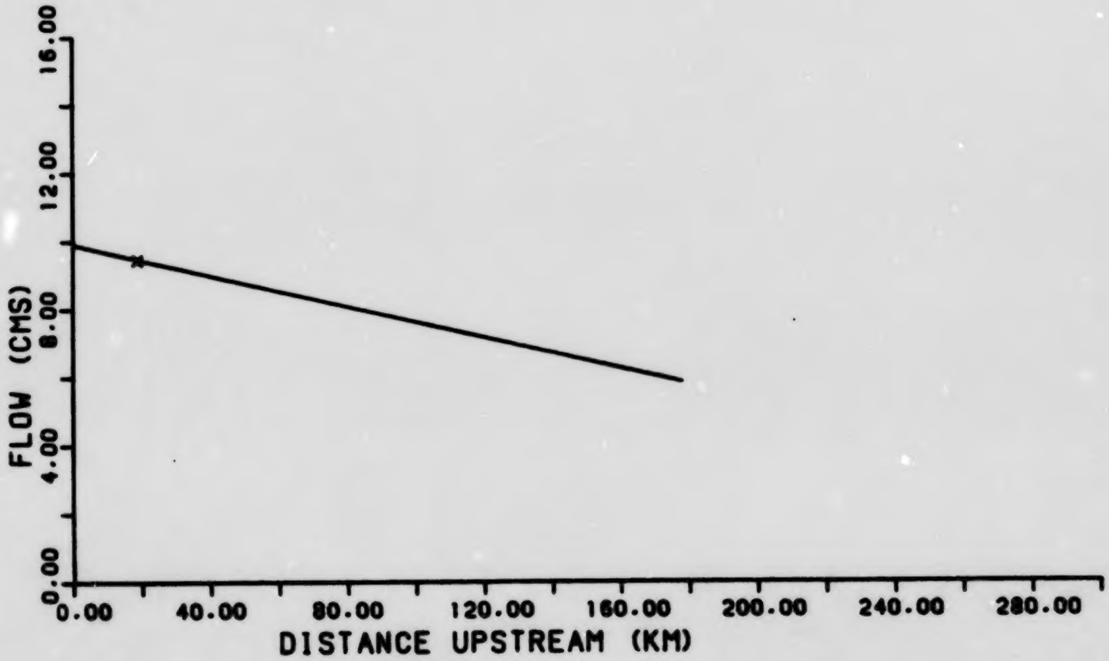
APPENDIX B

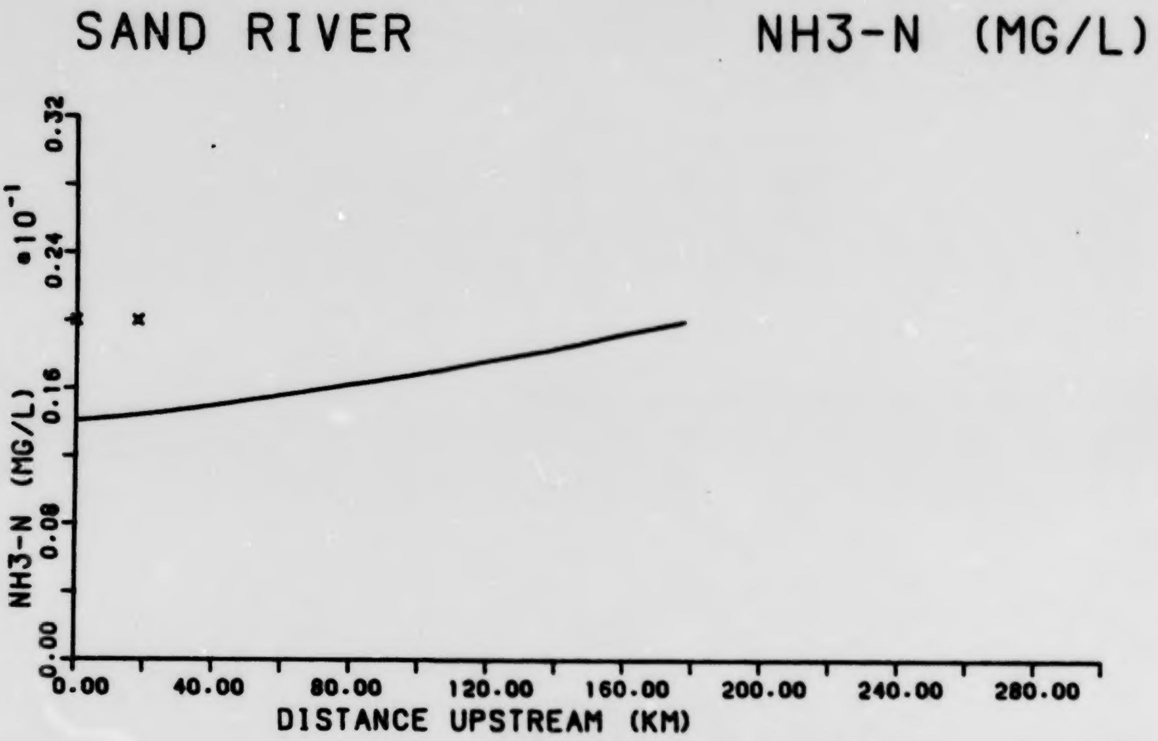
SAND RIVER CALIBRATION RESULTS

SAND RIVER DO (MG/L)

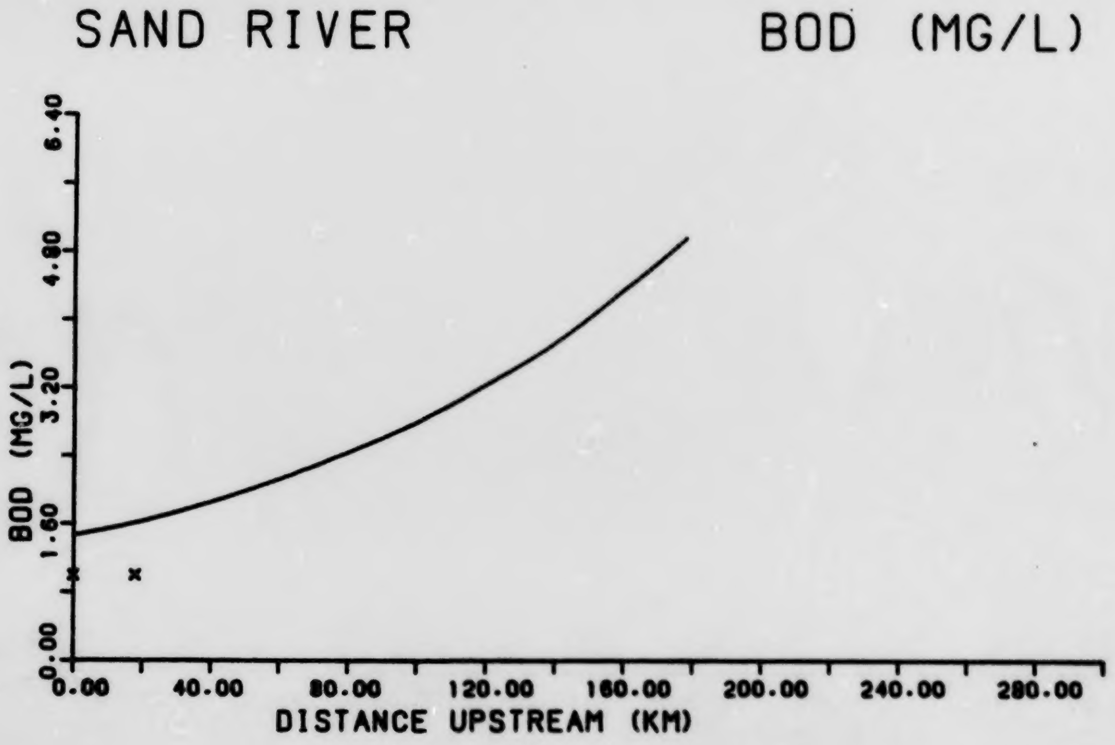


SAND RIVER FLOW (CMS)



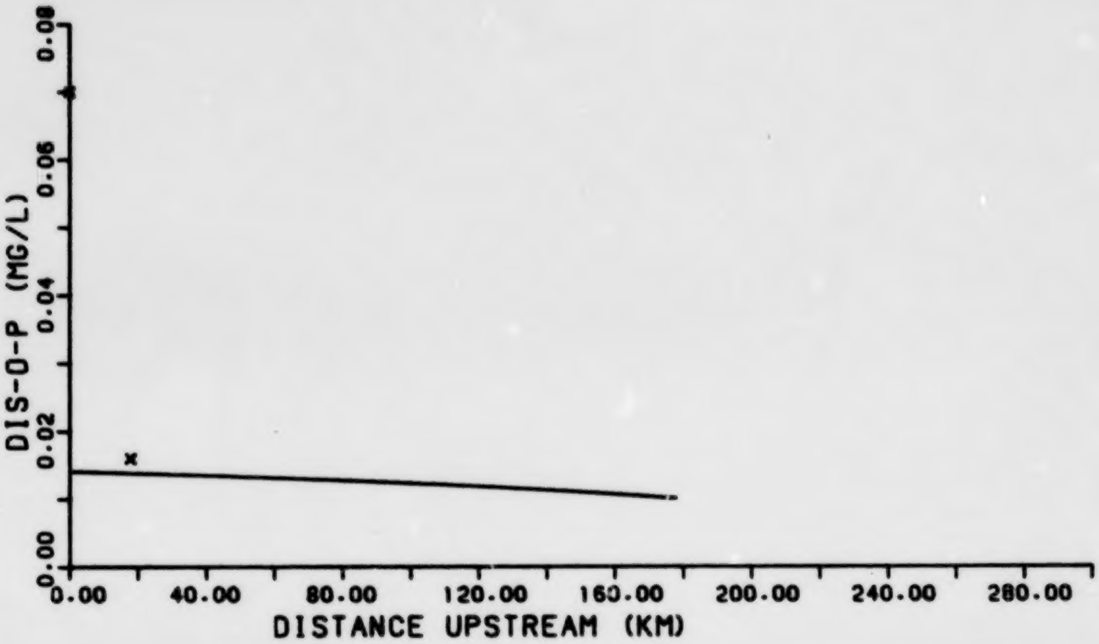


SAND RIVER NH3-N (MG/L)

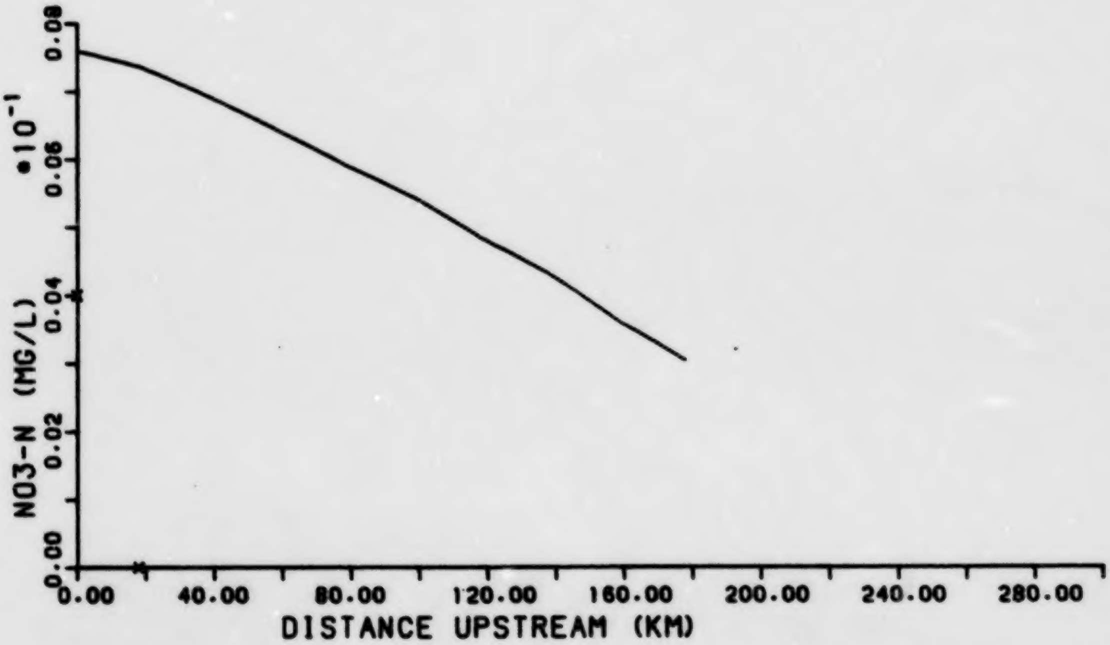


SAND RIVER BOD (MG/L)

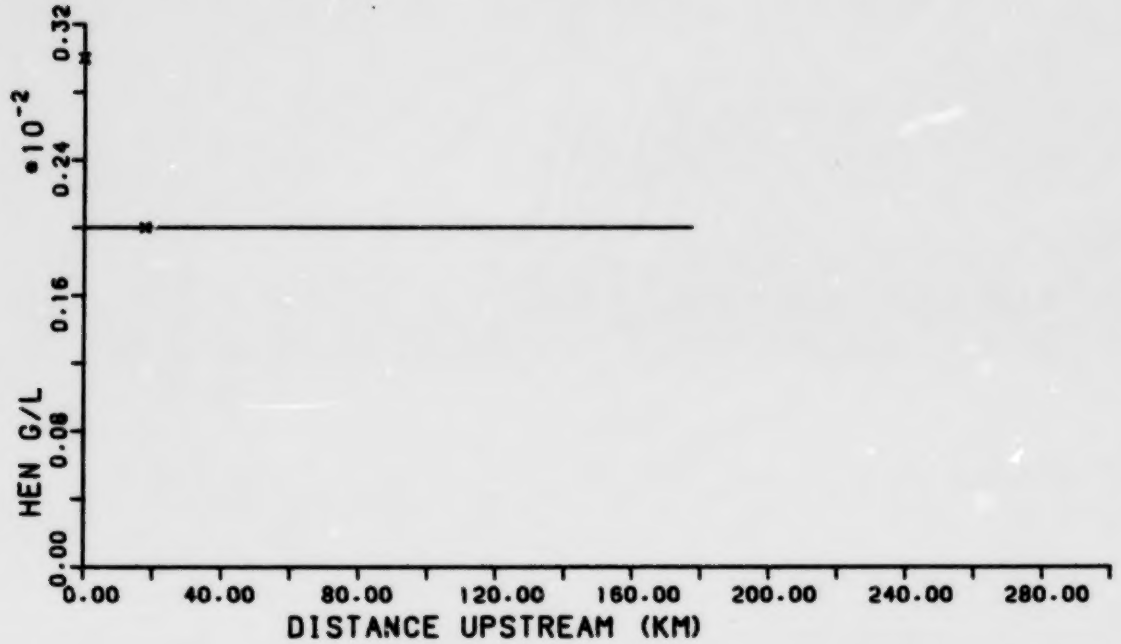
SAND RIVER DIS-O-P (MG/L)



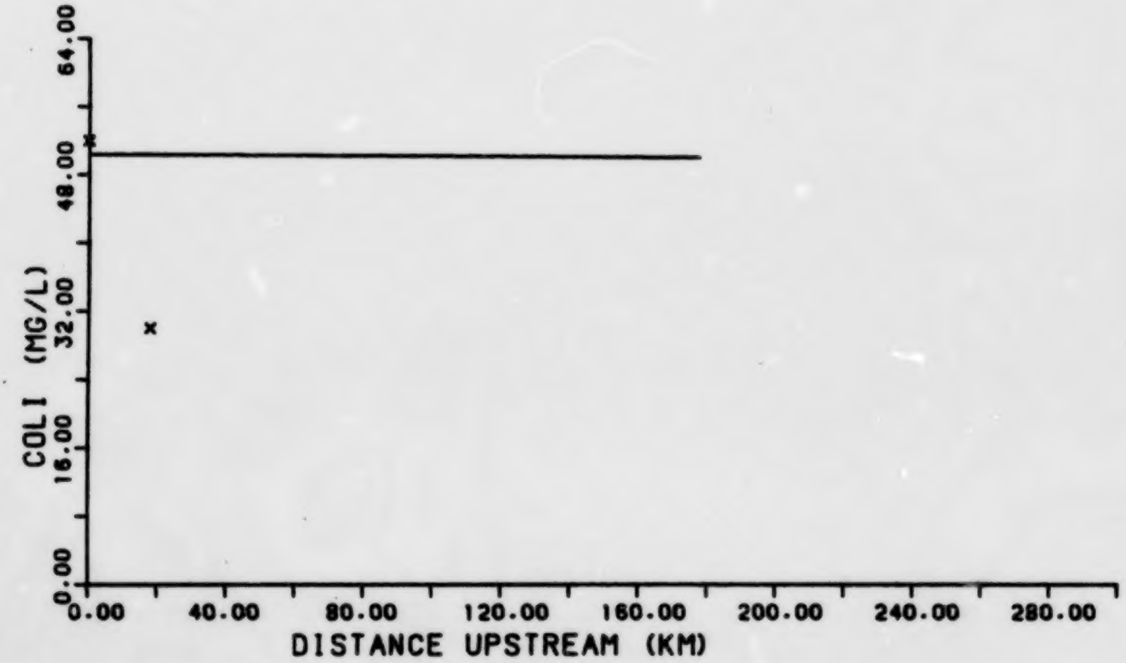
SAND RIVER NO3-N (MG/L)



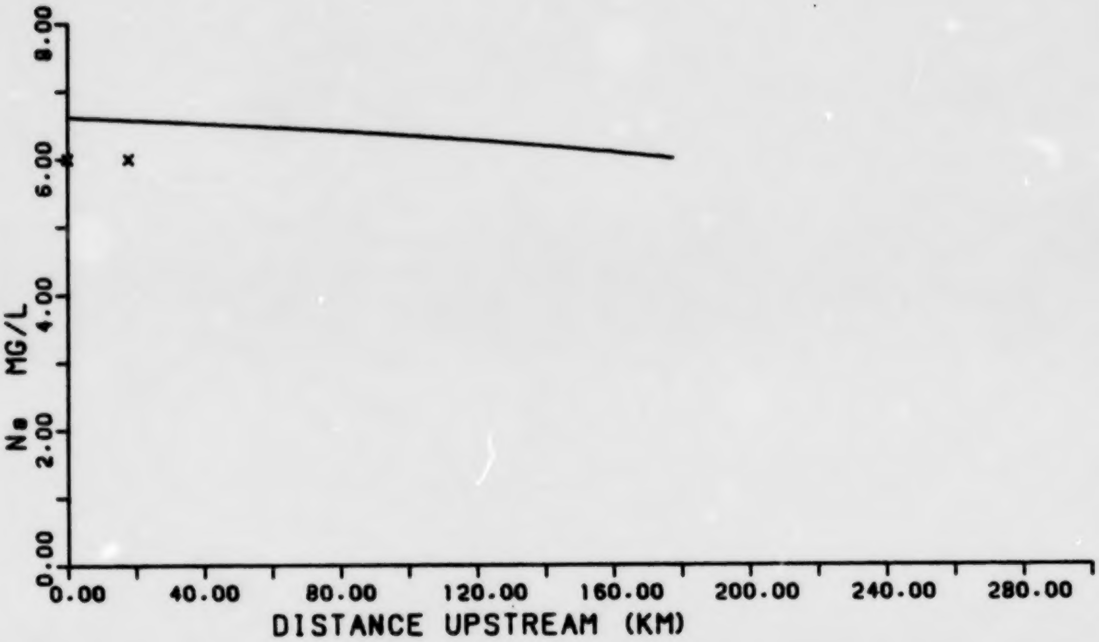
SAND RIVER HEN G/L



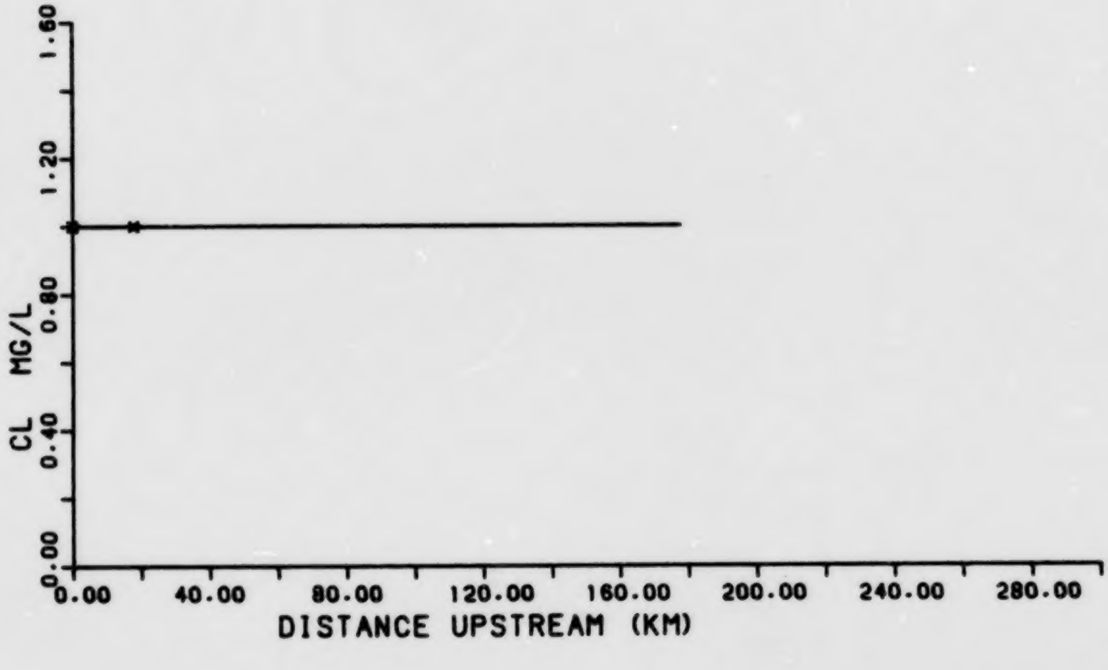
SAND RIVER COLI (MG/L)

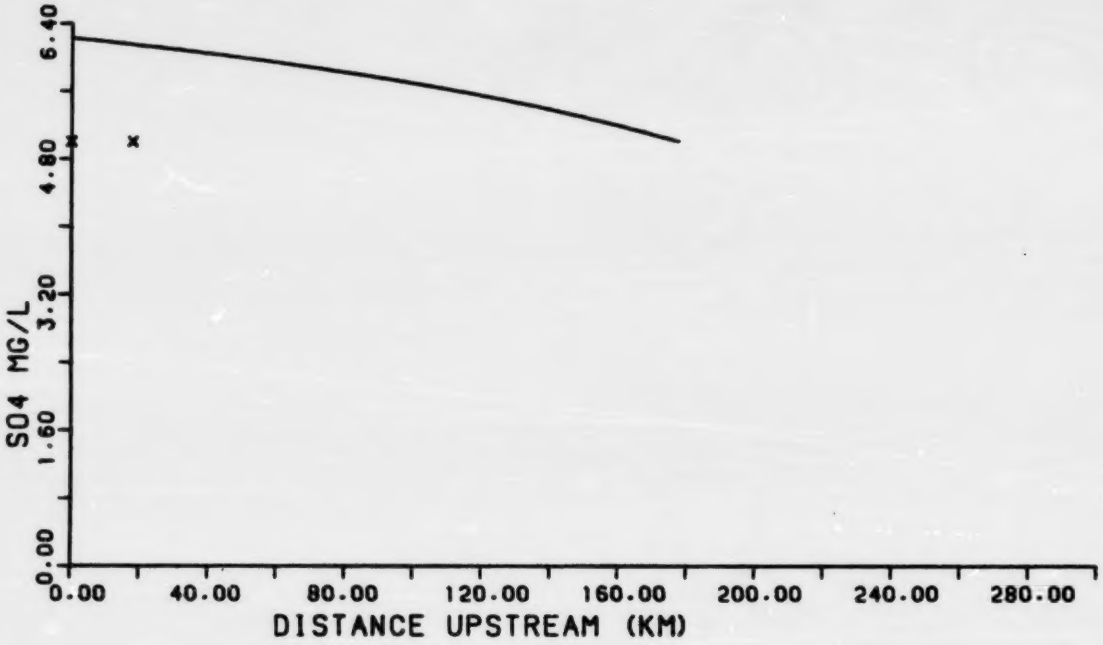


SAND RIVER Na MG/L



SAND RIVER CL MG/L





SAND RIVER SO4 MG/L

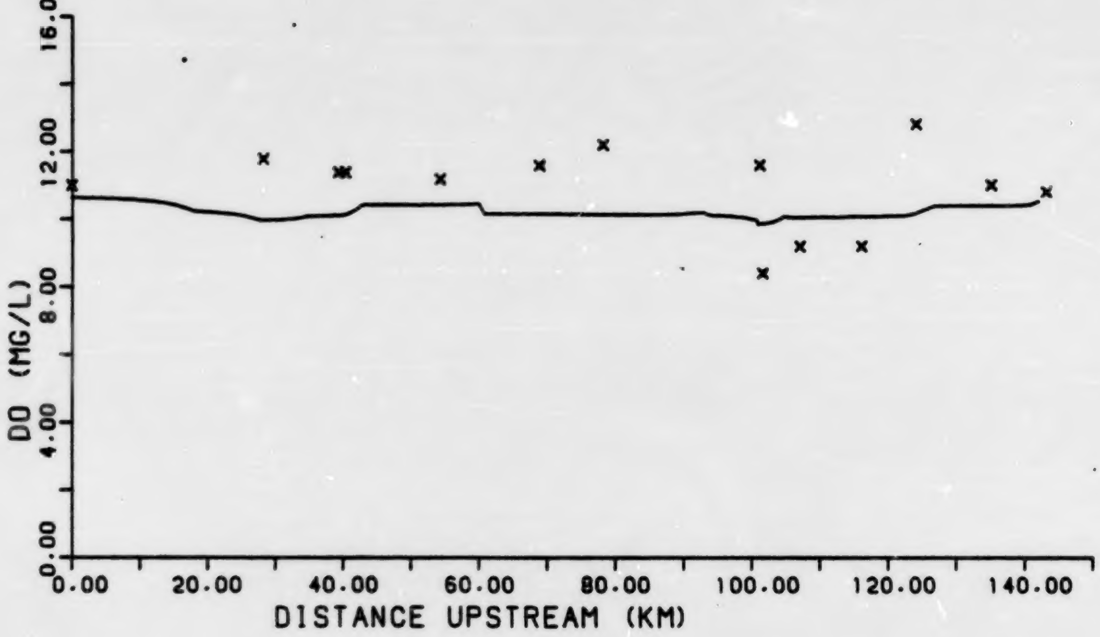
APPENDIX C

MODEL VERIFICATION RESULTS

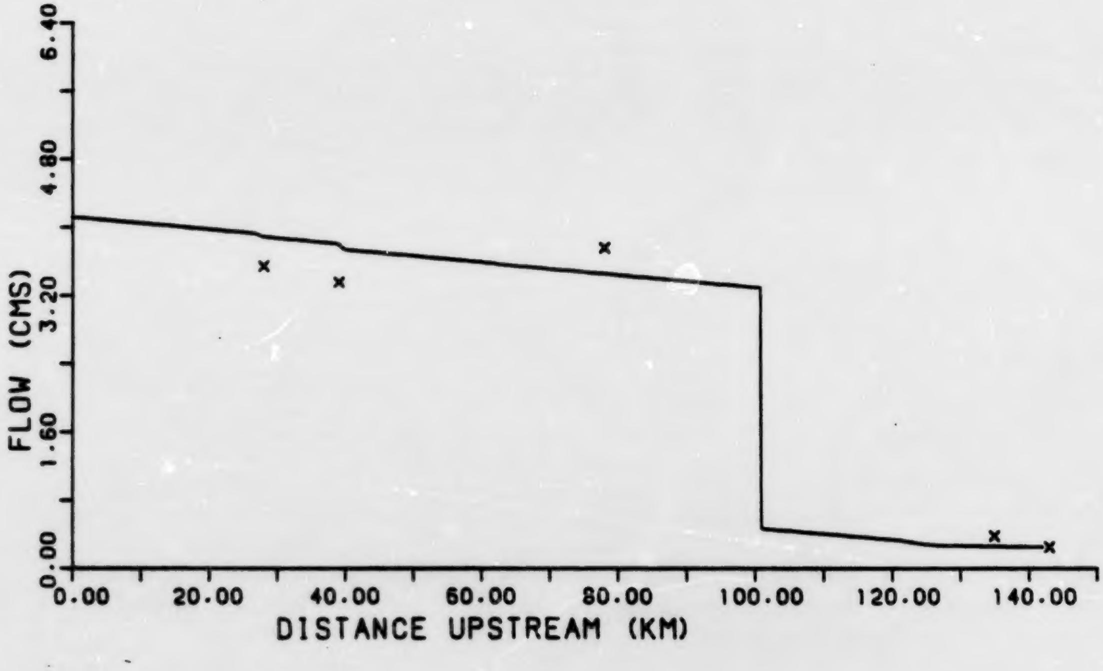
BEAVER RIVER

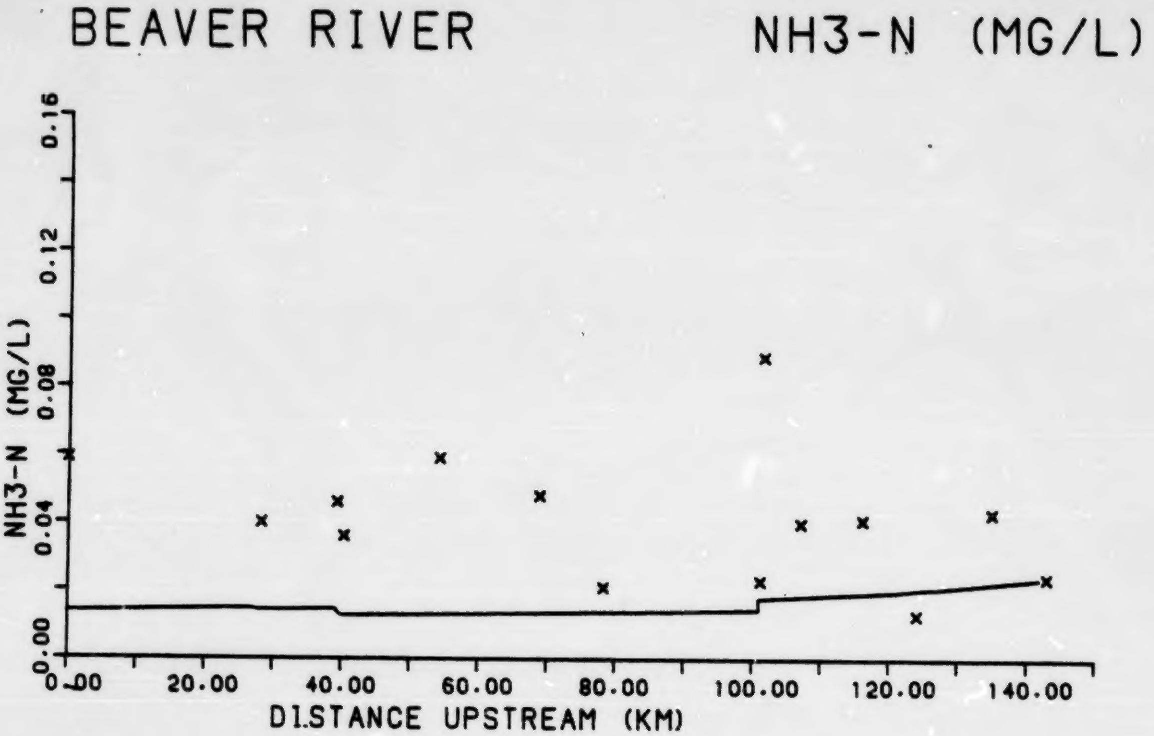
VERIFICATION RESULTS

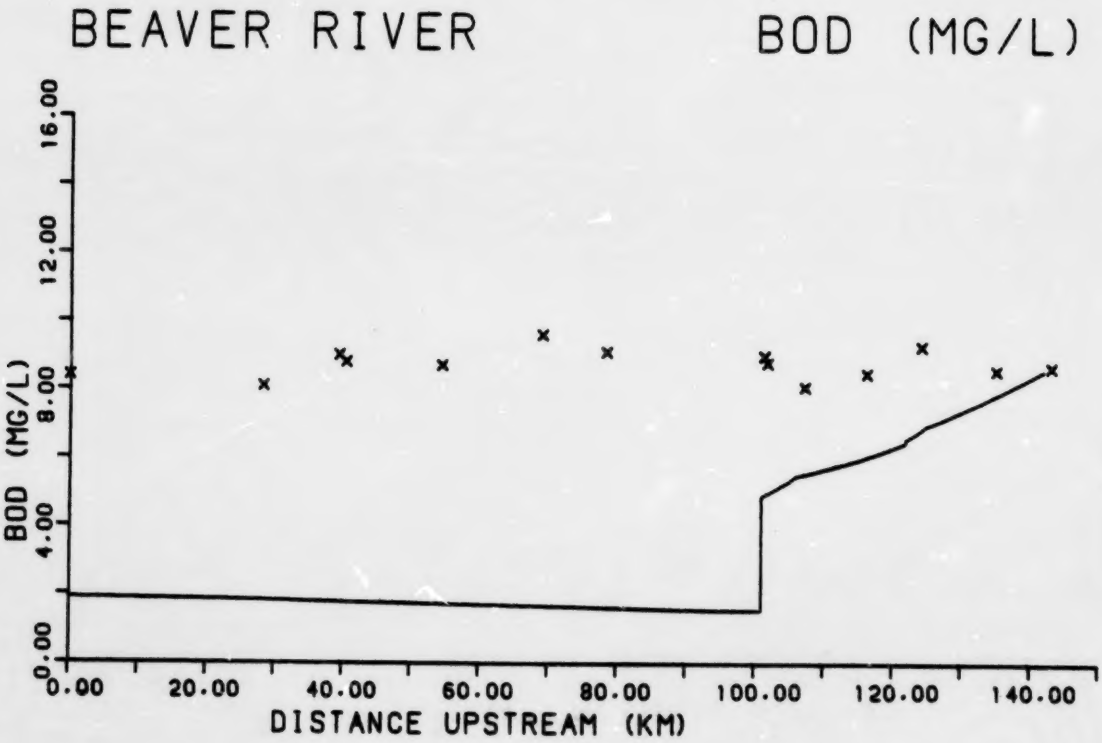
BEAVER RIVER DO (MG/L)

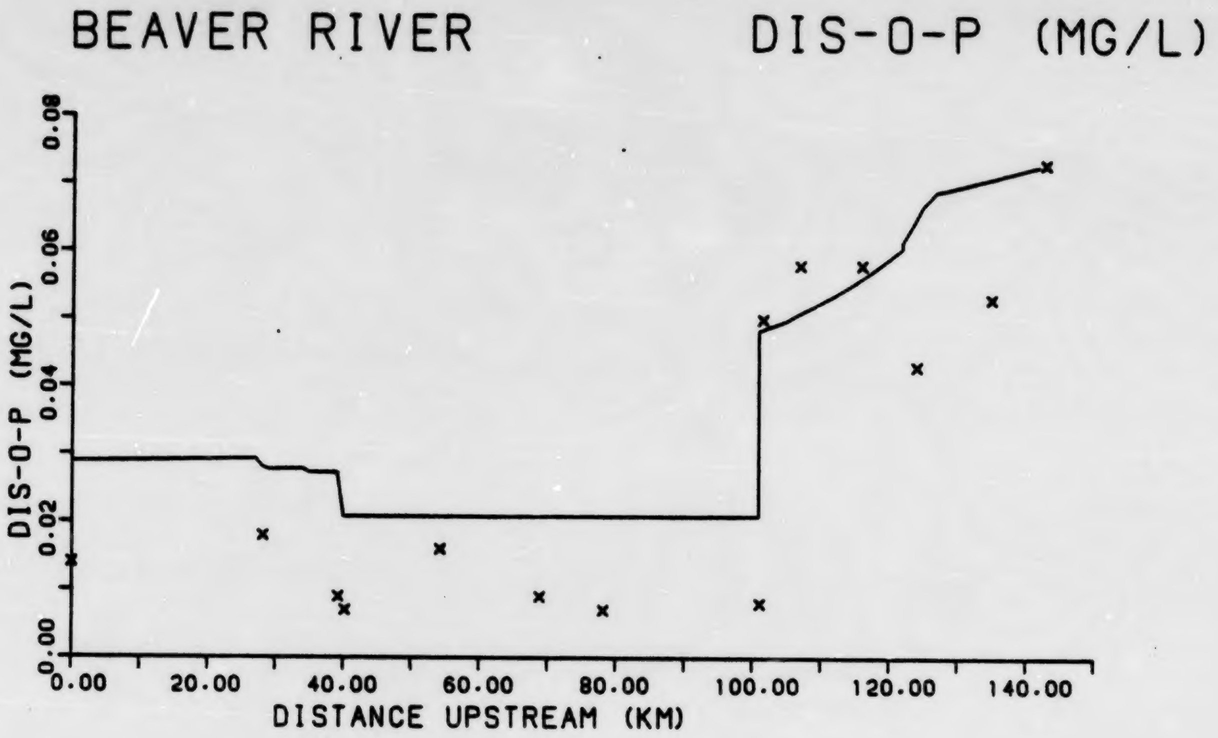


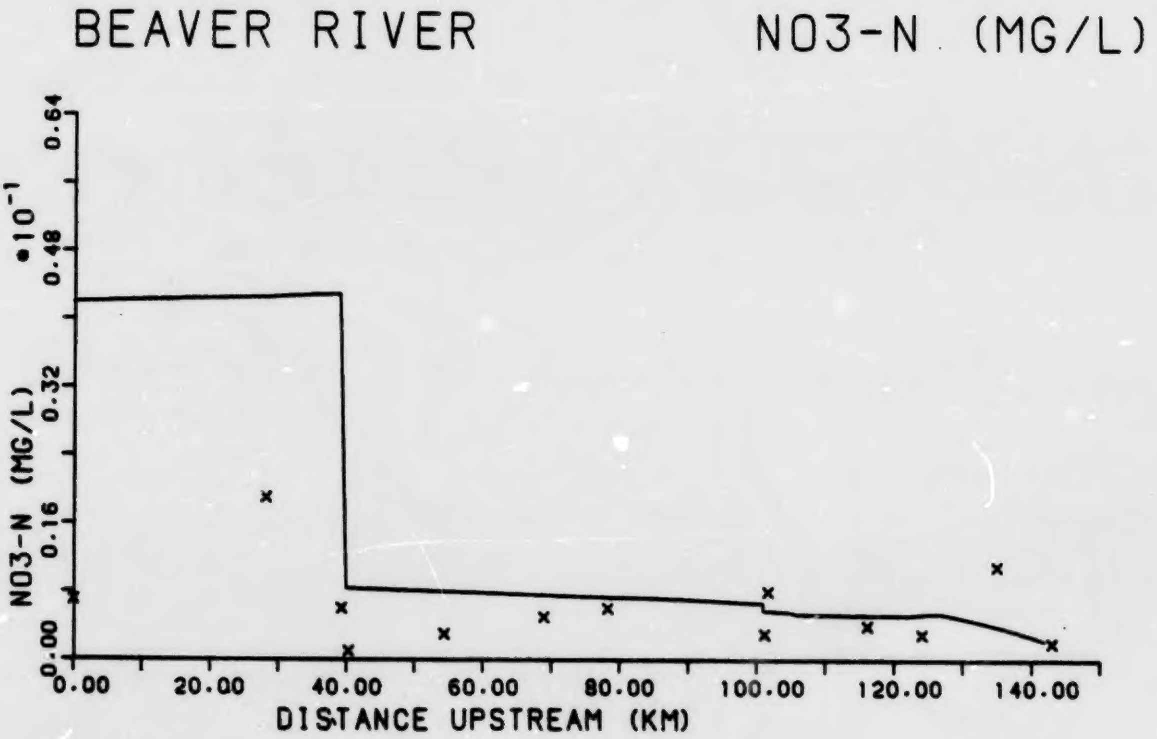
BEAVER RIVER FLOW (CMS)

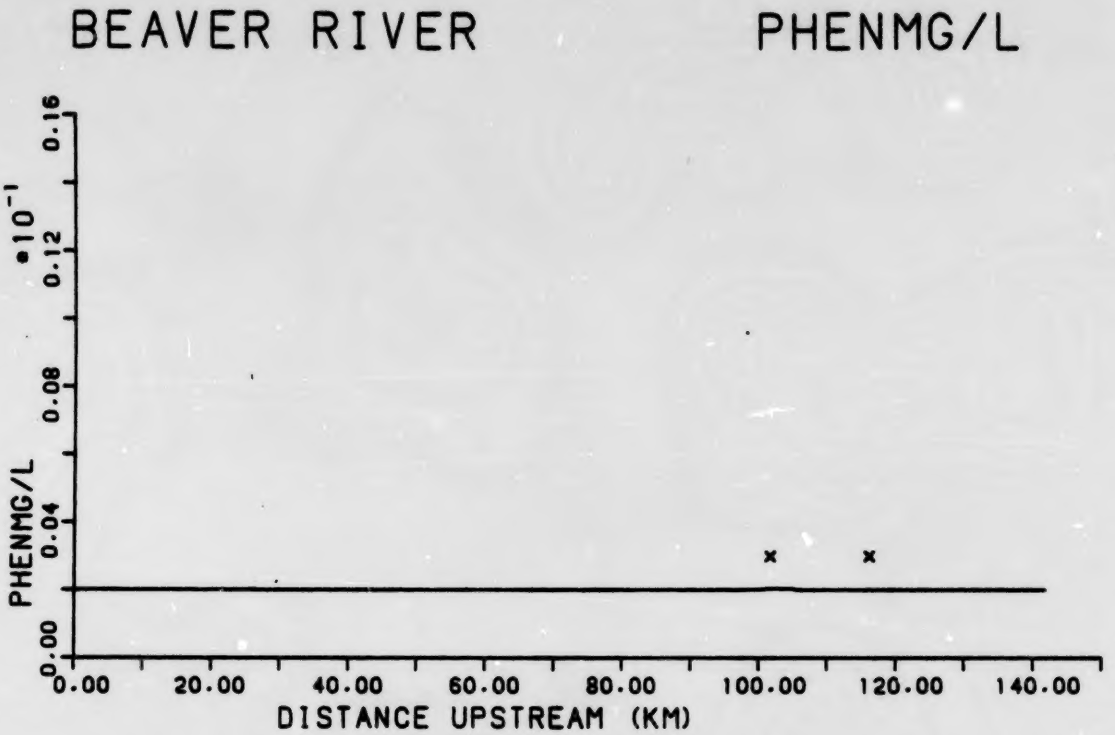


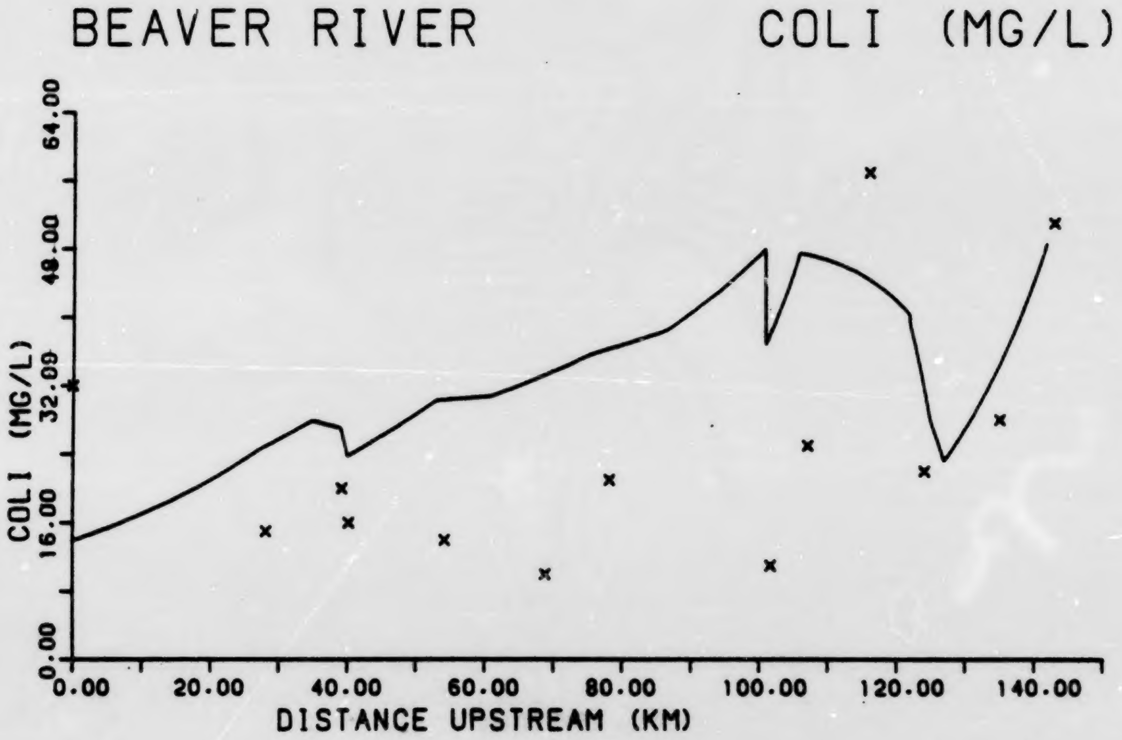

BEAVER RIVER
NH3-N (MG/L)
NH3-N (MG/L)
0.00
0.04
0.08
0.12
0.16
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

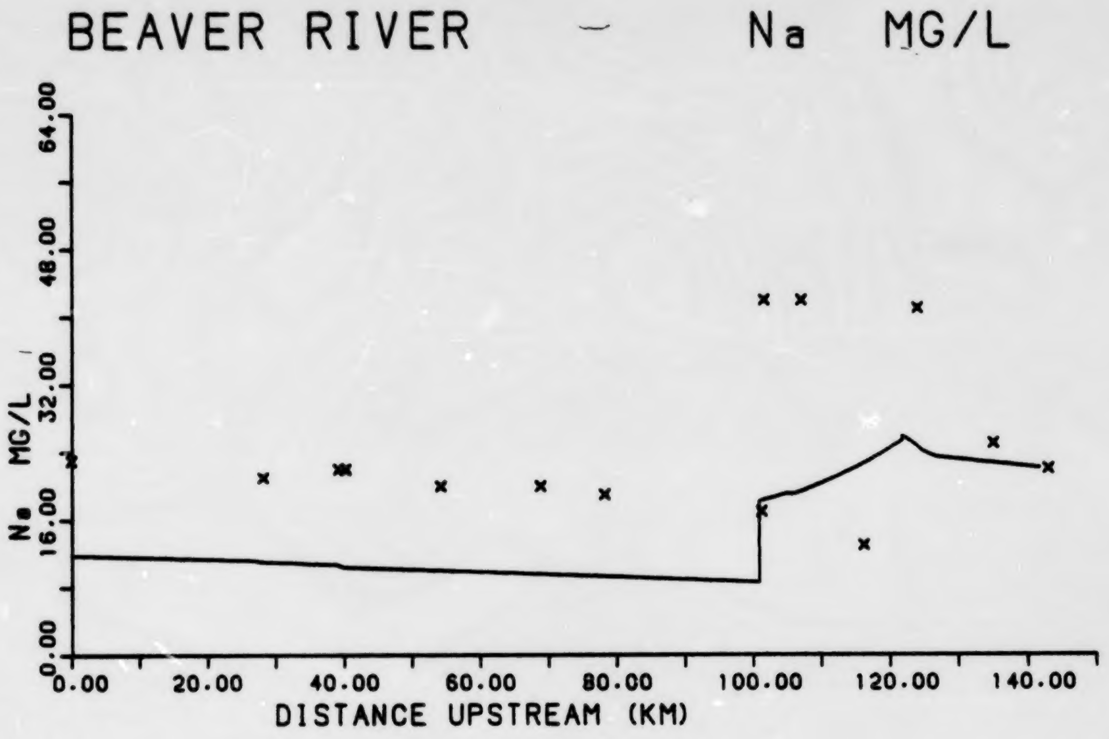

BEAVER RIVER
BOD (MG/L)
BOD (MG/L)
0.00
4.00
8.00
12.00
16.00
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

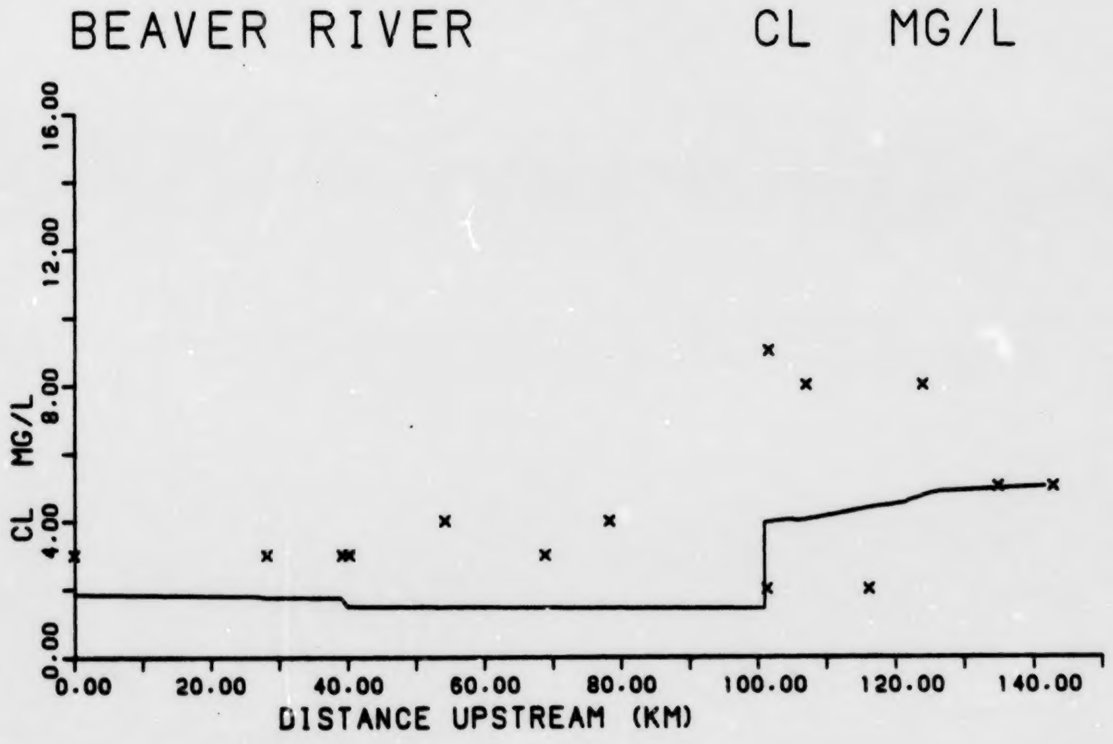

BEAVER RIVER
DIS-O-P (MG/L)
DIS-O-P (MG/L)
0.00
0.02
0.04
0.06
0.08
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

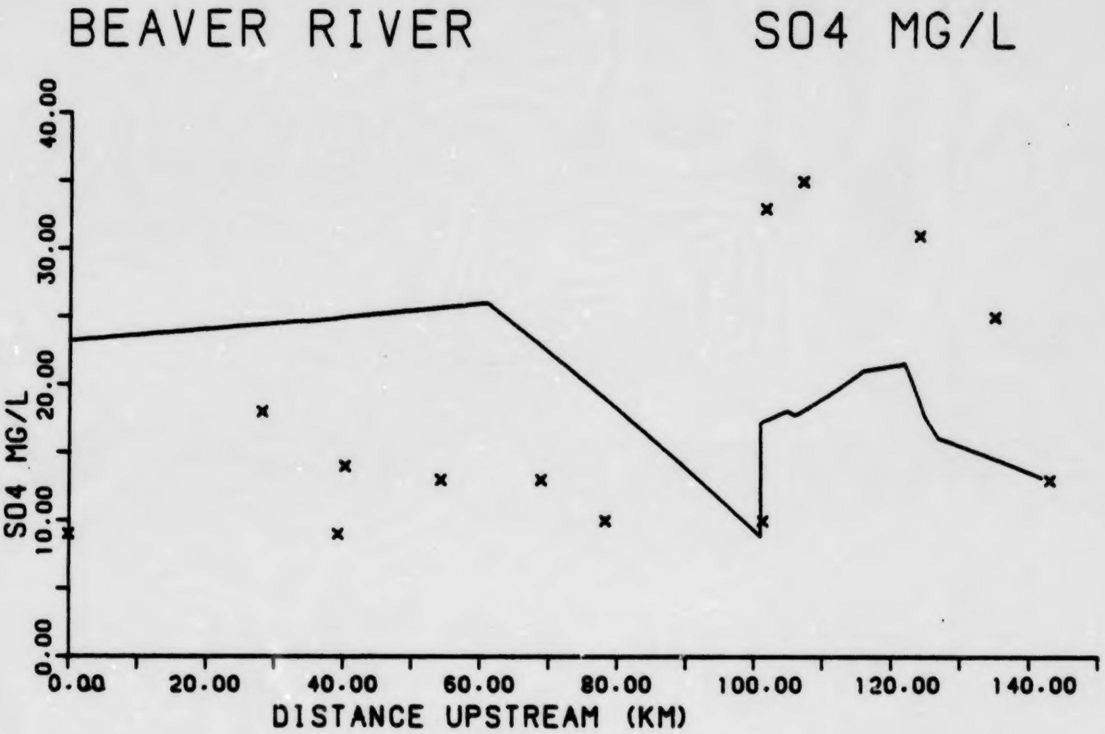

BEAVER RIVER
NO3-N (MG/L)
NO3-N (MG/L)
*10⁻¹
0.00
0.16
0.32
0.48
0.64
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)


BEAVER RIVER
PHENMG/L
PHENMG/L
*10⁻¹
0.00
0.04
0.08
0.12
0.16
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)


BEAVER RIVER
COLI (MG/L)
COLI (MG/L)
0.00
16.00
32.00
48.00
64.00
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)


BEAVER RIVER — Na MG/L
Na MG/L
0.00
16.00
32.00
48.00
64.00
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)


BEAVER RIVER CL MG/L
CL MG/L
0.00
4.00
8.00
12.00
16.00
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

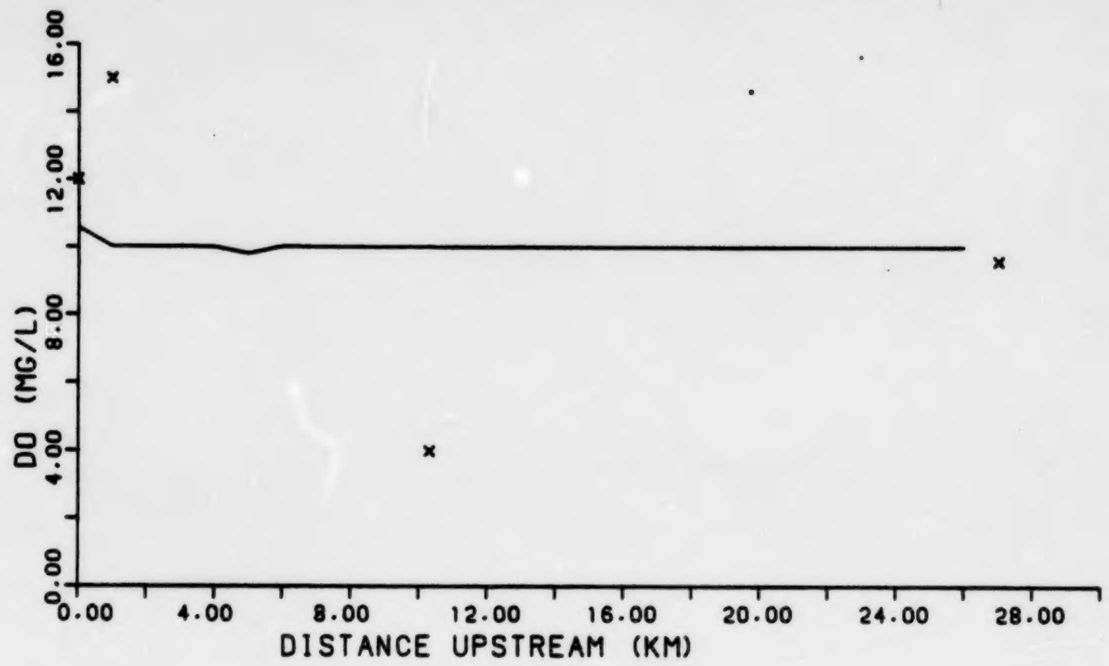


BEAVER RIVER SO4 MG/L

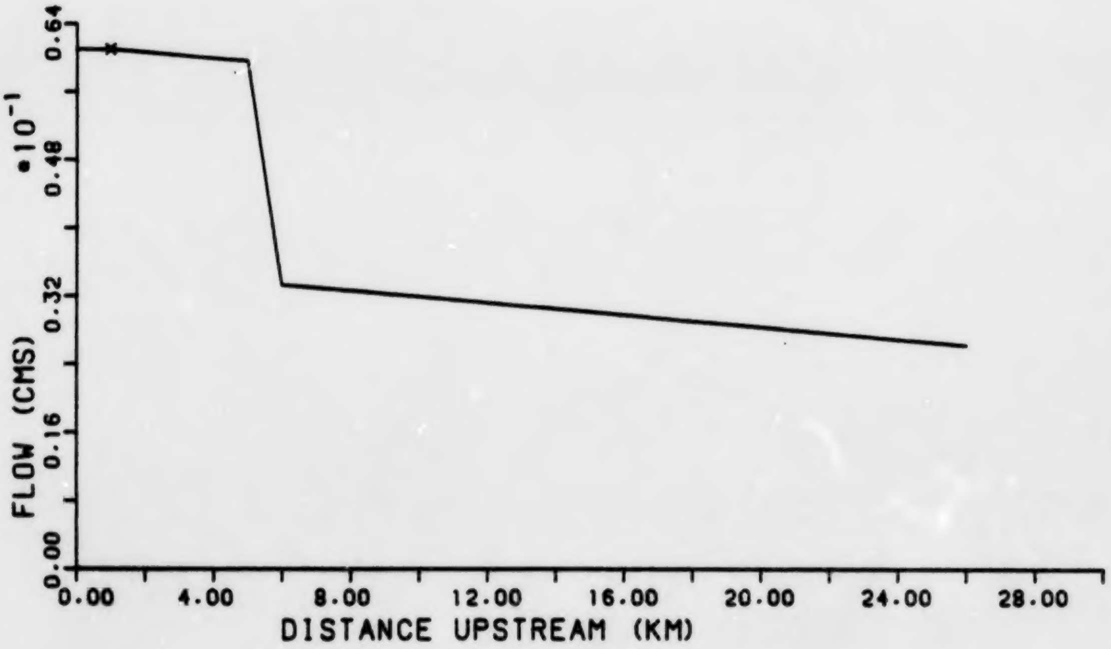
MARIE CREEK

VERIFICATION RESULTS

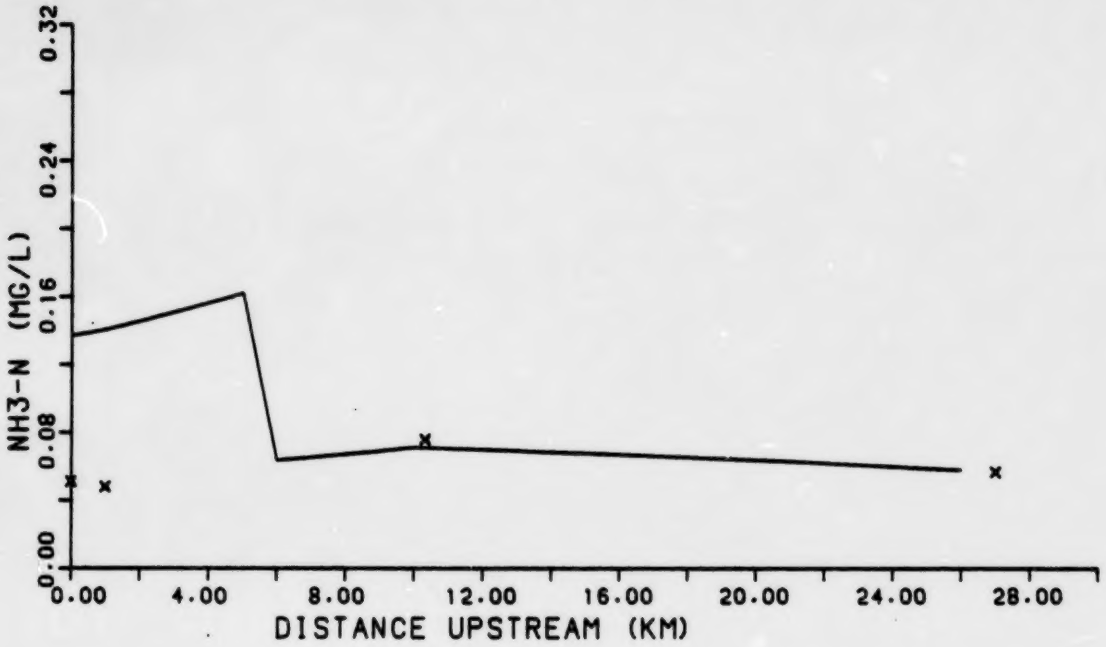
MARIE CREEK DO (MG/L)



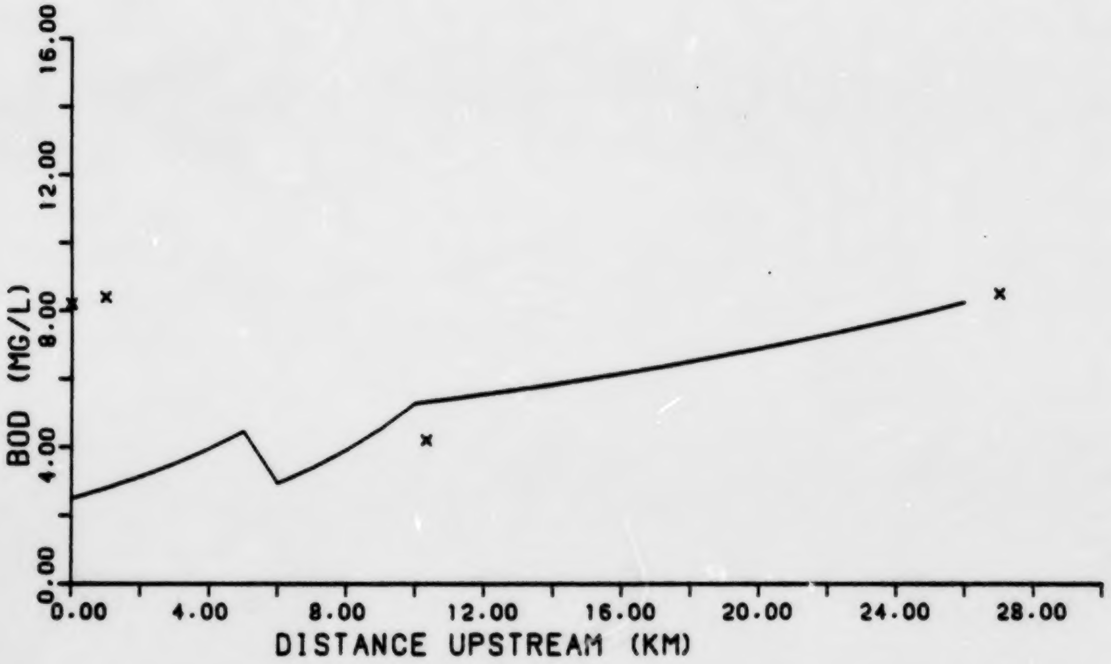
MARIE CREEK FLOW (CMS)



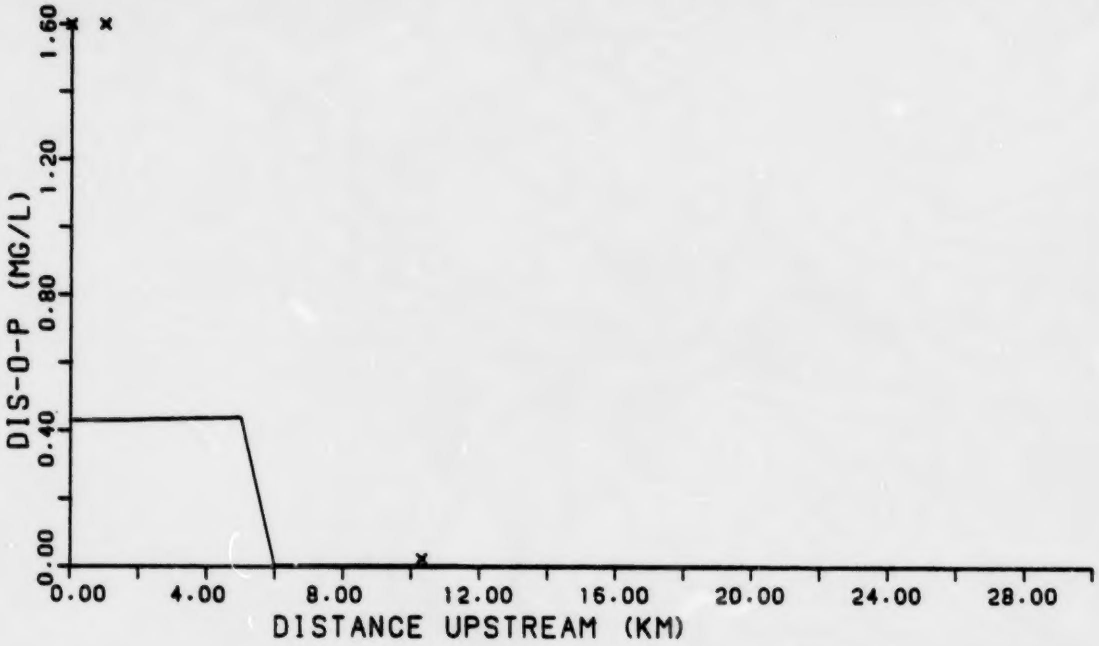
MARIE CREEK NH3-N (MG/L)



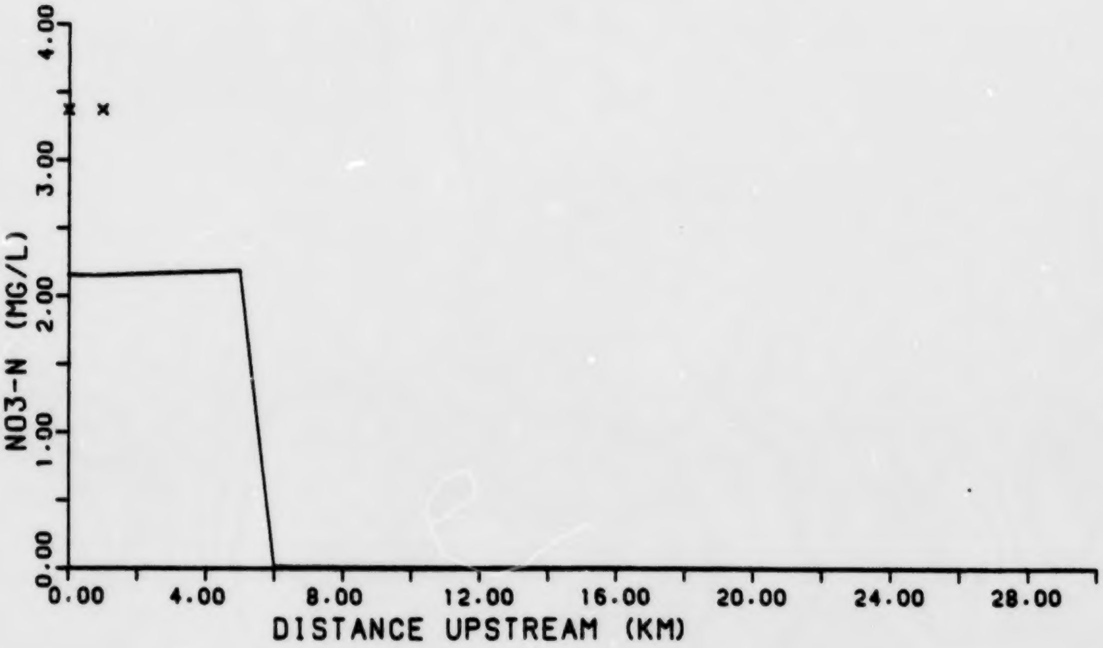
MARIE CREEK BOD (MG/L)



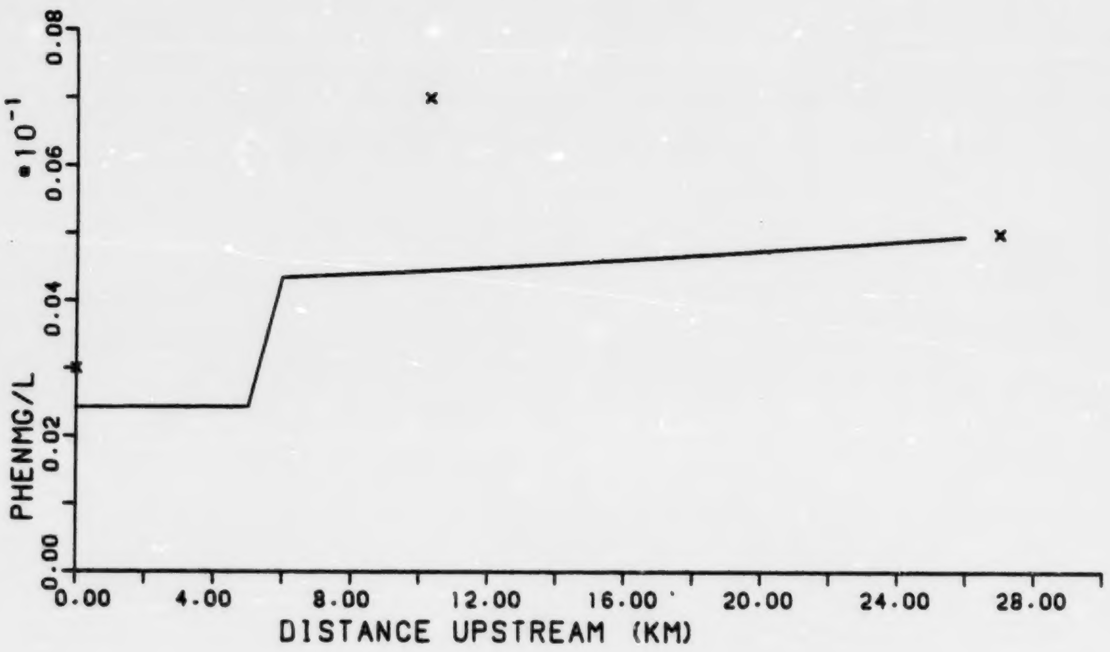
MARIE CREEK DIS-O-P (MG/L)



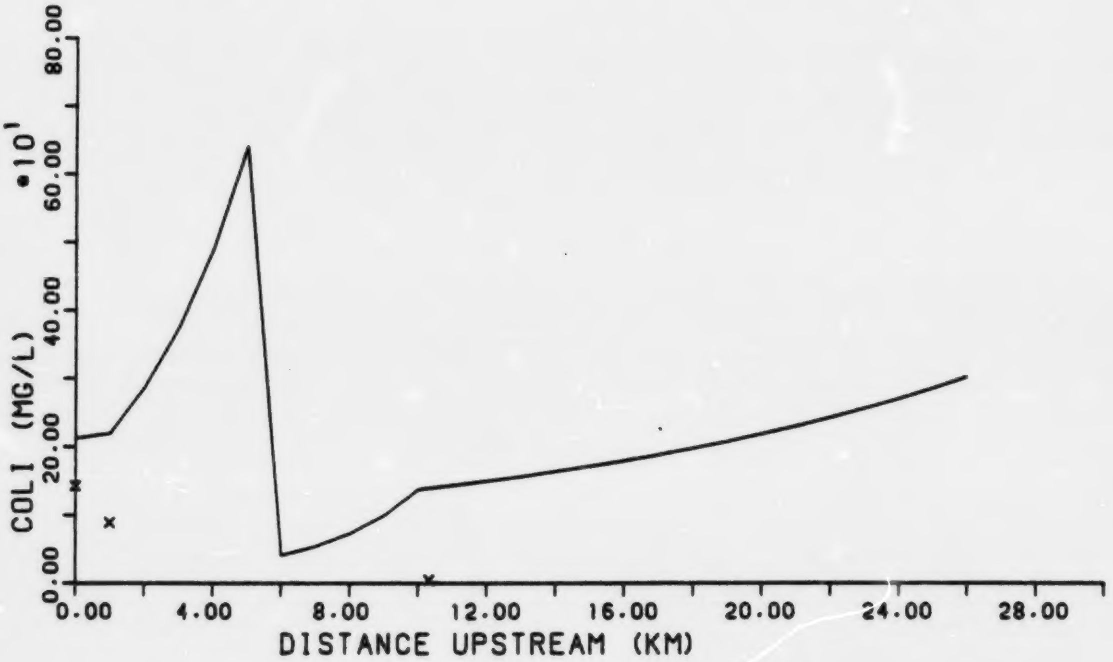
MARIE CREEK NO3-N (MG/L)

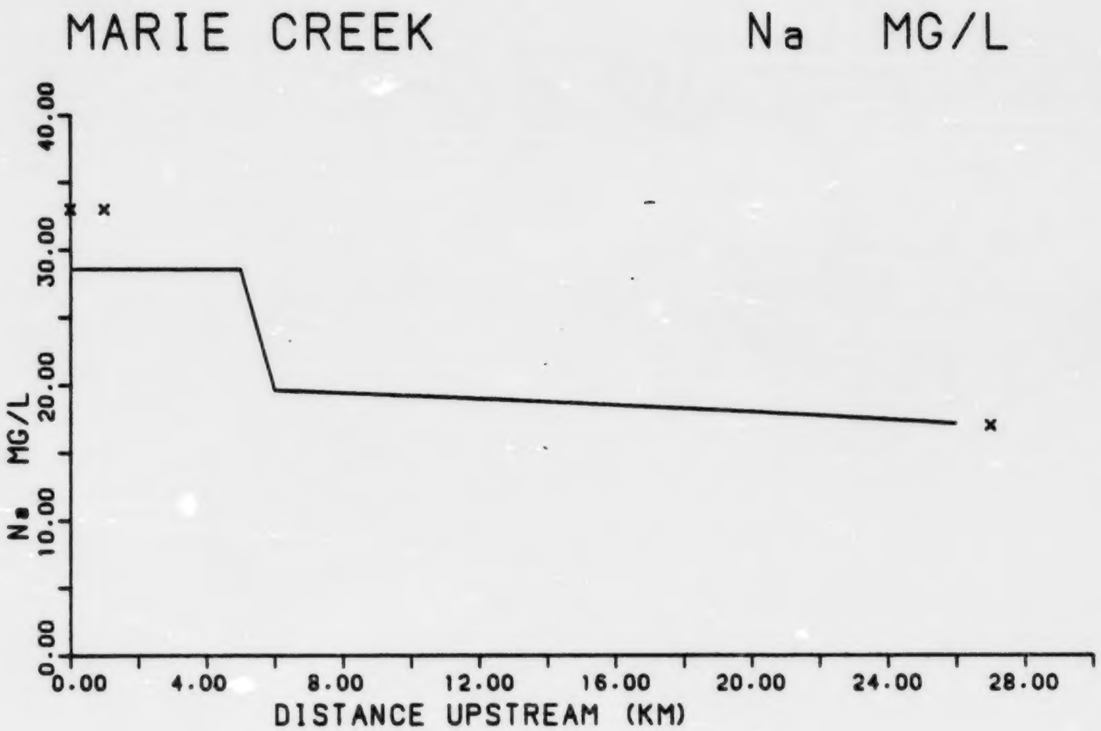


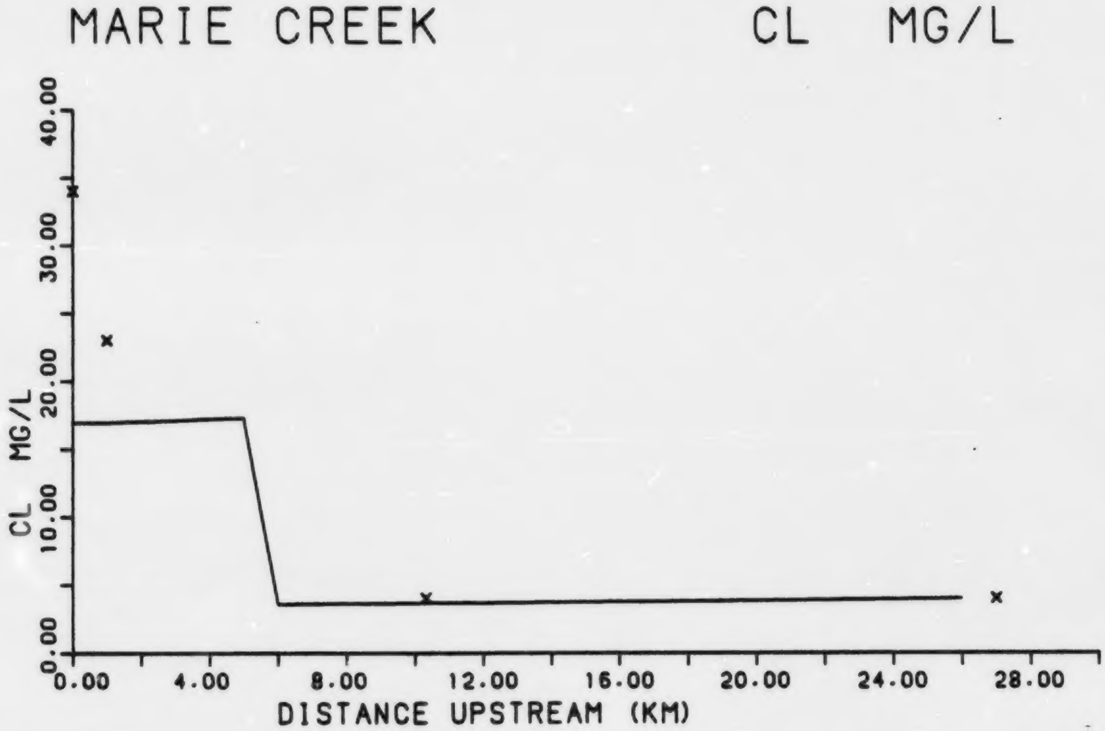
MARIE CREEK PHENMG/L

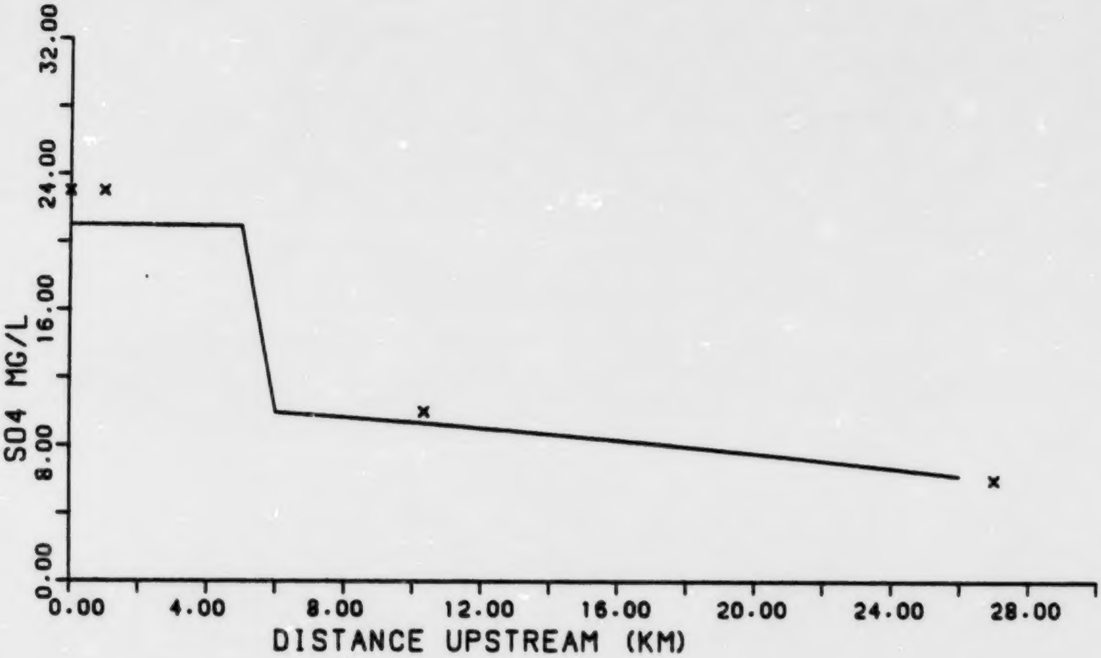


MARIE CREEK COLI (MG/L)




MARIE CREEK
Na MG/L
Na MG/L
0.00
10.00
20.00
30.00
40.00
0.00
4.00
8.00
12.00
16.00
20.00
24.00
28.00
DISTANCE UPSTREAM (KM)


MARIE CREEK
CL MG/L
CL MG/L
0.00
10.00
20.00
30.00
40.00
0.00
4.00
8.00
12.00
16.00
20.00
24.00
28.00
DISTANCE UPSTREAM (KM)



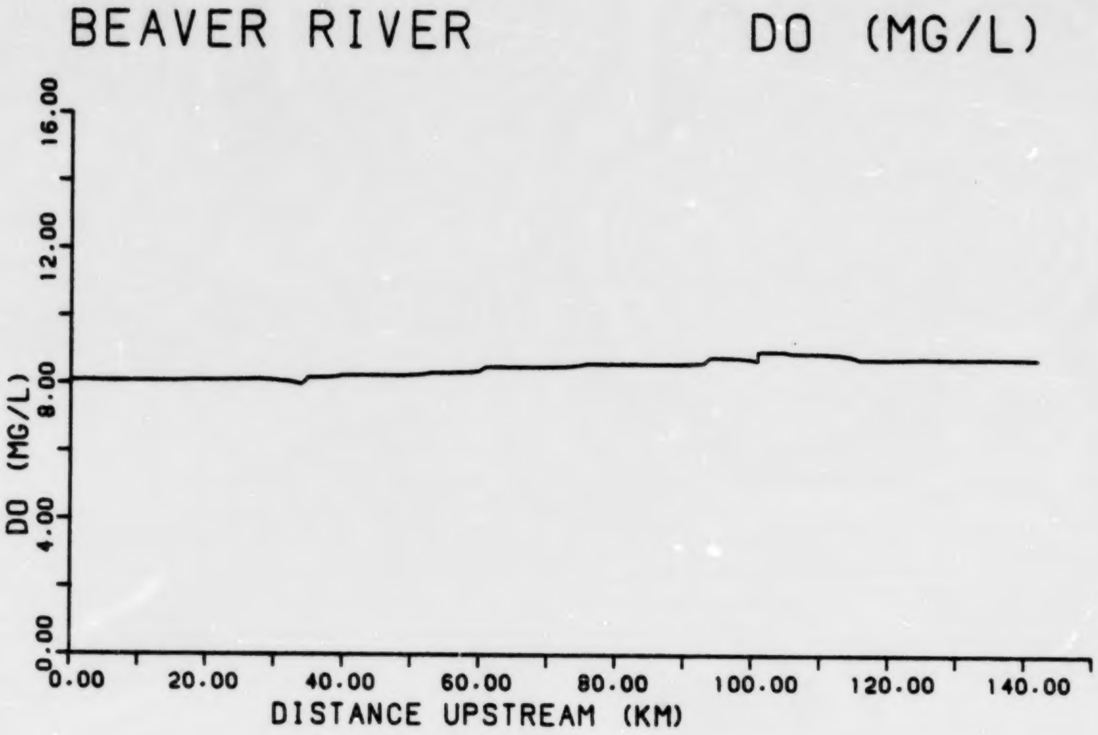
MARIE CREEK SO4 MG/L

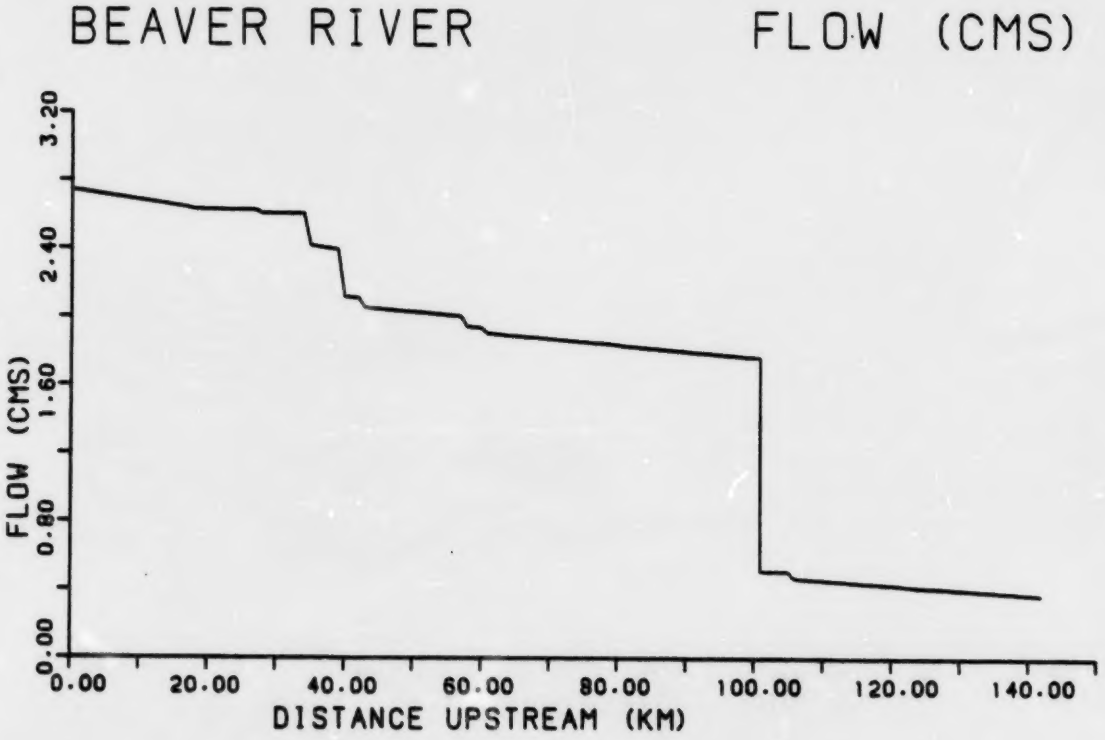
APPENDIX D

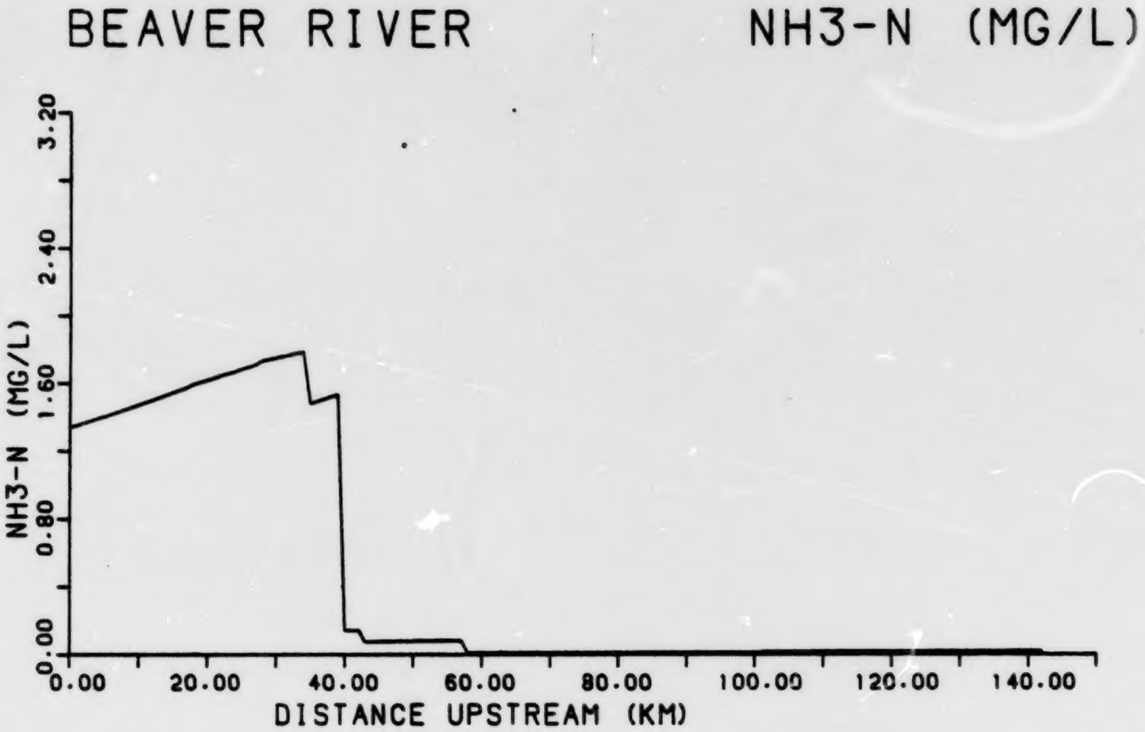
PREDICTED WATER
QUALITY, 2000 AD

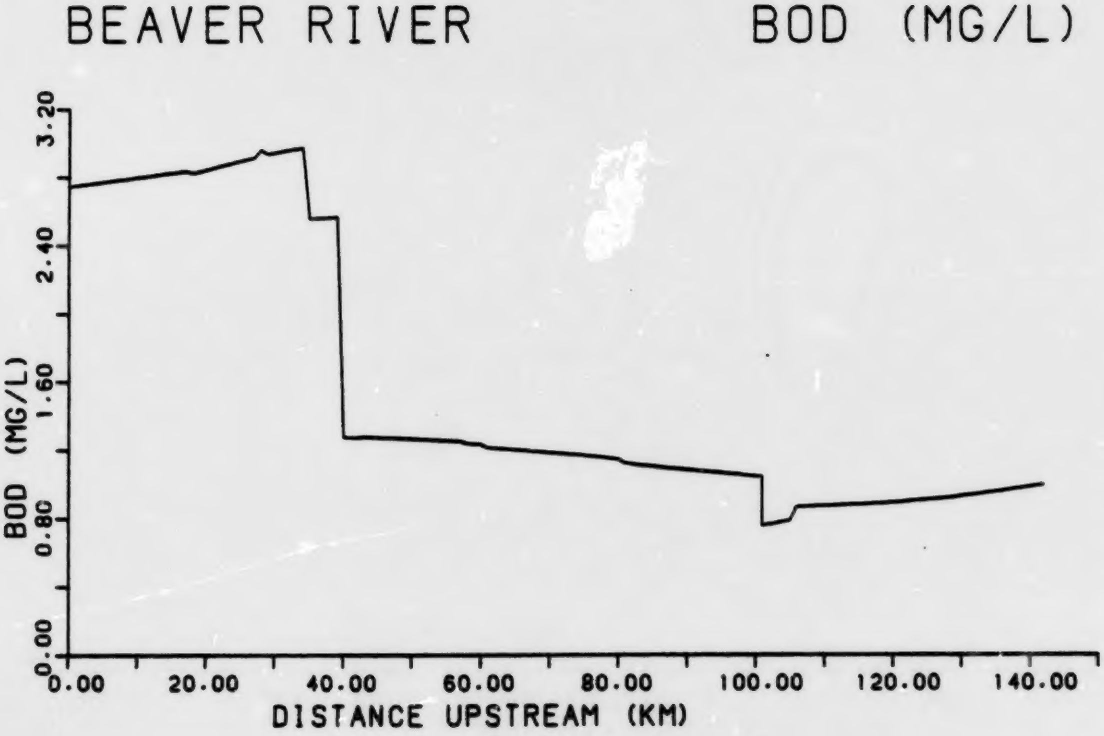
PRODUCTION RUN NO. 1,
SUMMER LOW FLOW

BEAVER RIVER

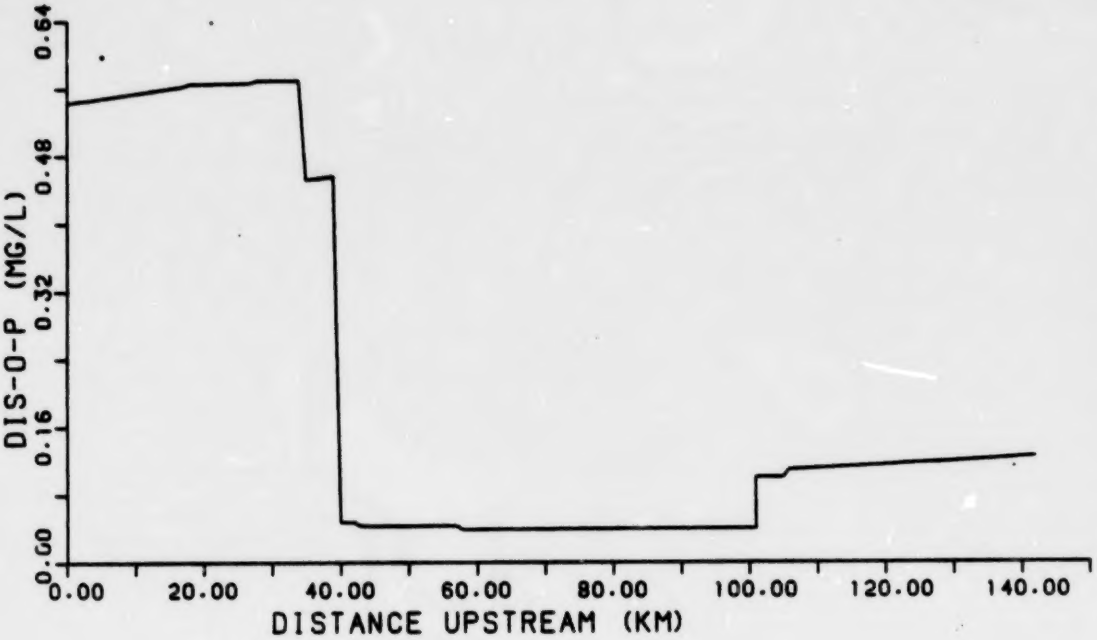

BEAVER RIVER
DO (MG/L)
16.00
12.00
8.00
4.00
0.00
DO (MG/L)
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)


BEAVER RIVER
FLOW (CMS)
3.20
2.40
1.60
0.80
0.00
FLOW (CMS)
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

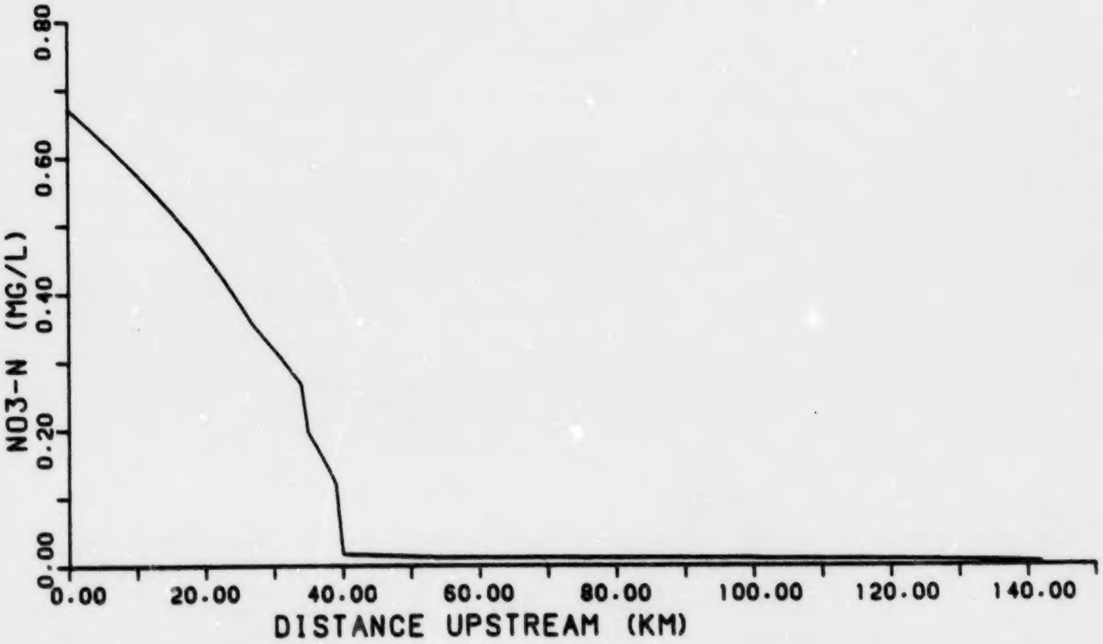

BEAVER RIVER
NH3-N (MG/L)
NH3-N (MG/L)
0.00
0.80
1.60
2.40
3.20
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

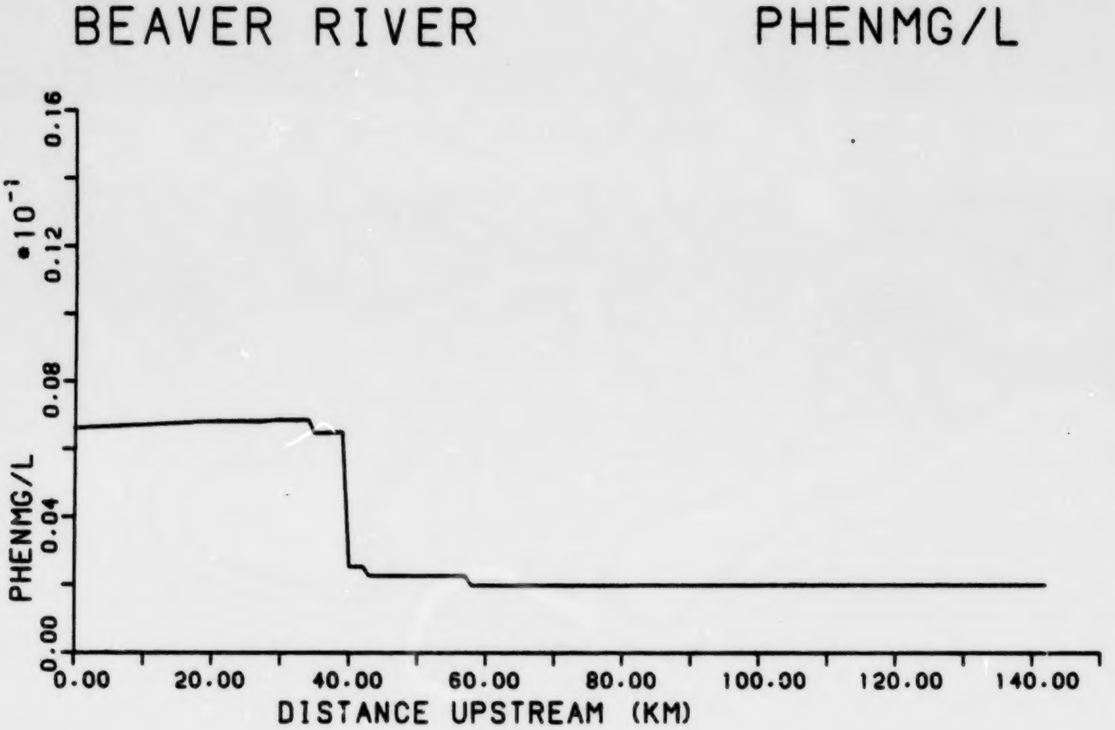

BEAVER RIVER
BOD (MG/L)
BOD (MG/L)
0.00
0.80
1.60
2.40
3.20
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

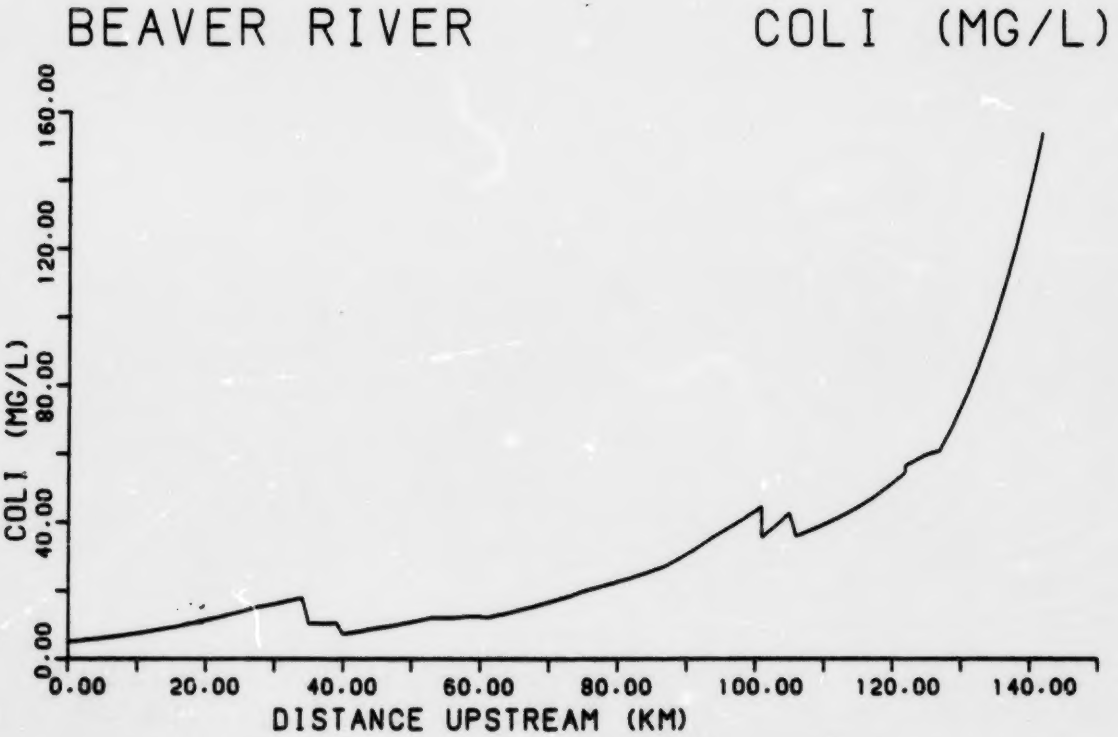
BEAVER RIVER DIS-O-P (MG/L)

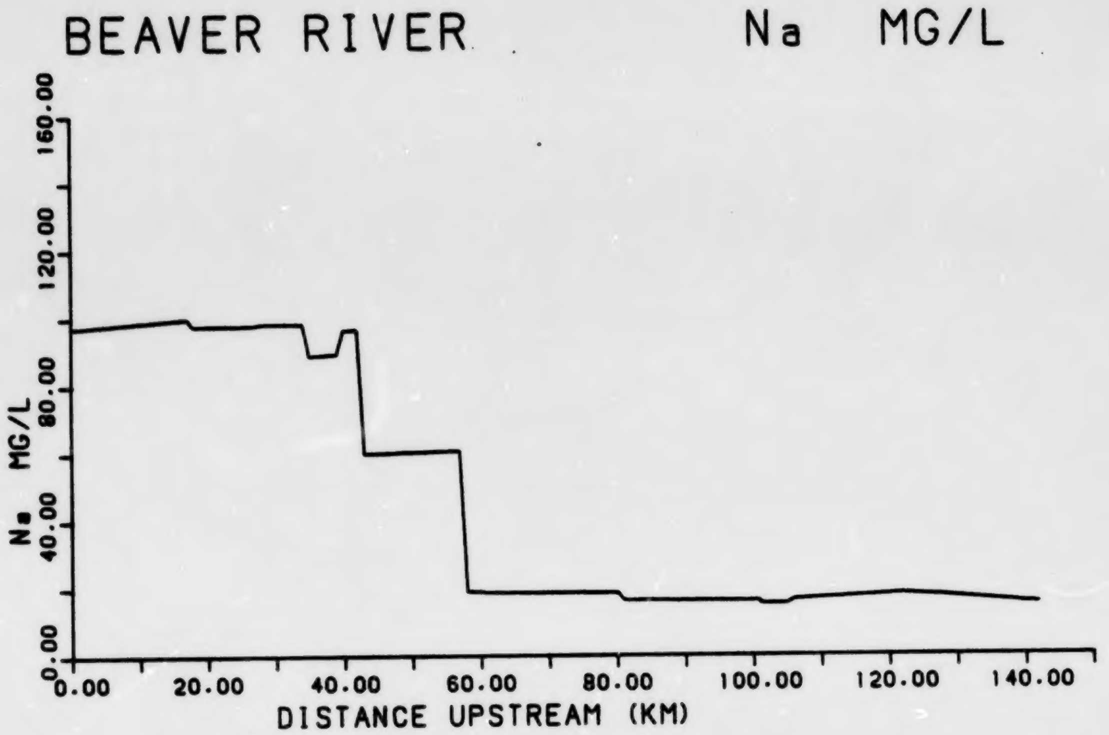


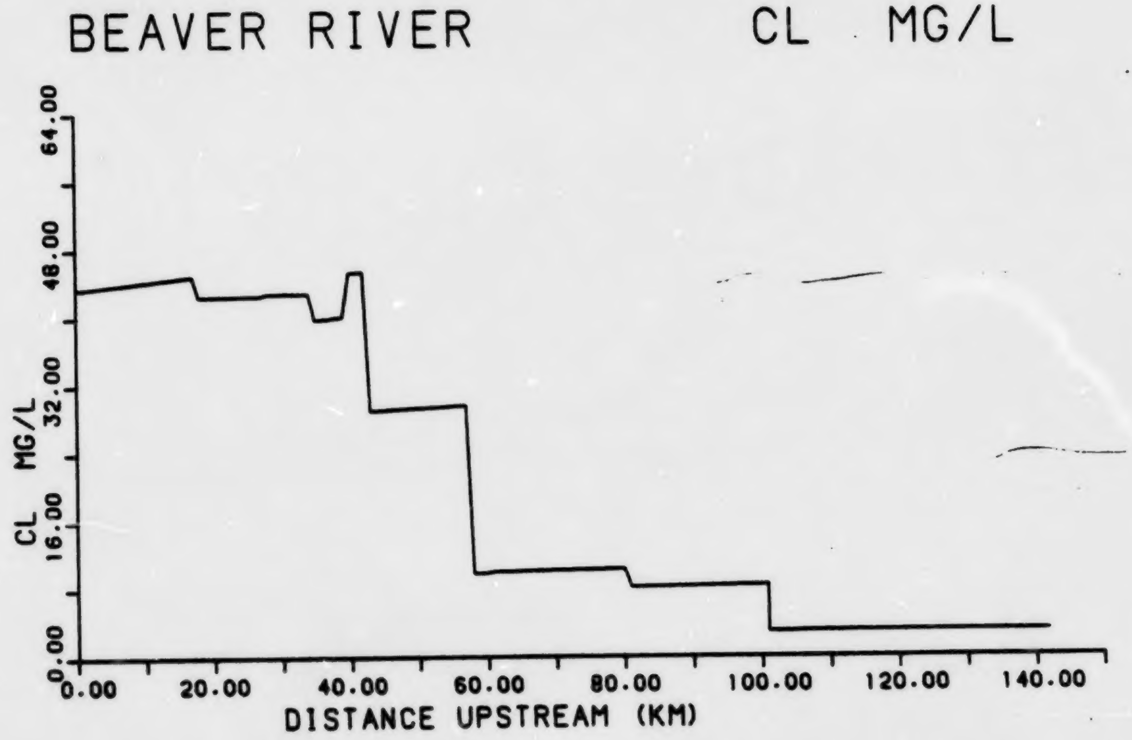
BEAVER RIVER NO3-N (MG/L)

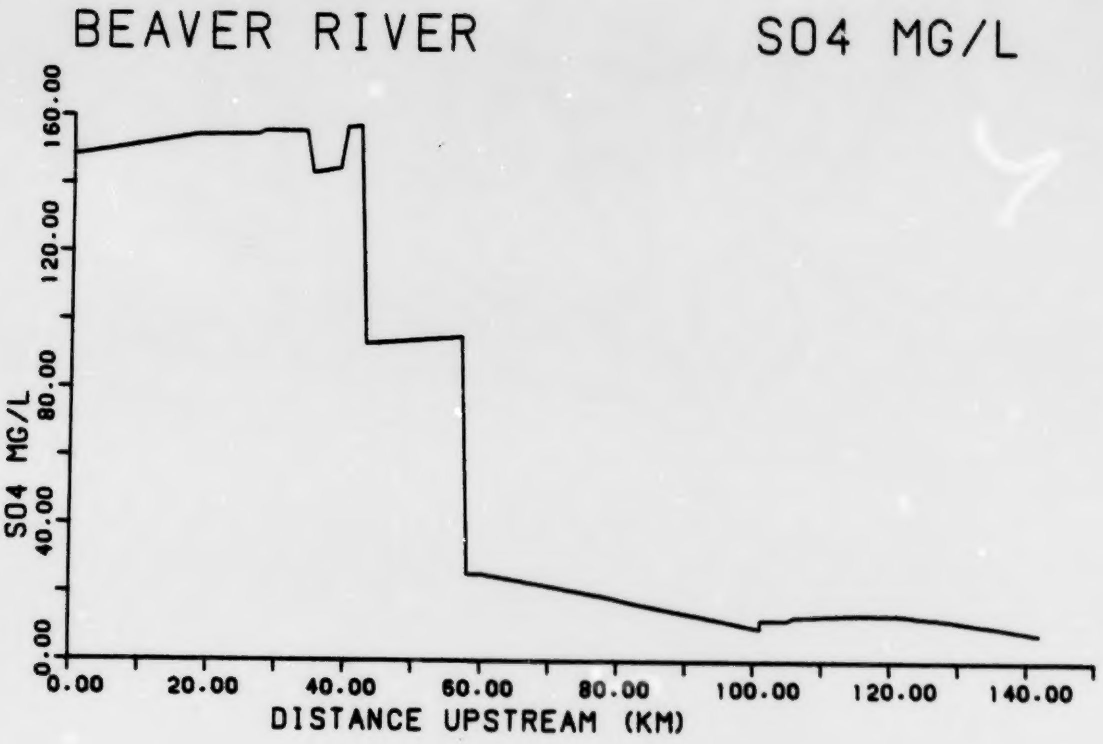



BEAVER RIVER
PHENMG/L
PHENMG/L
*10⁻¹
0.00
0.04
0.08
0.12
0.16
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

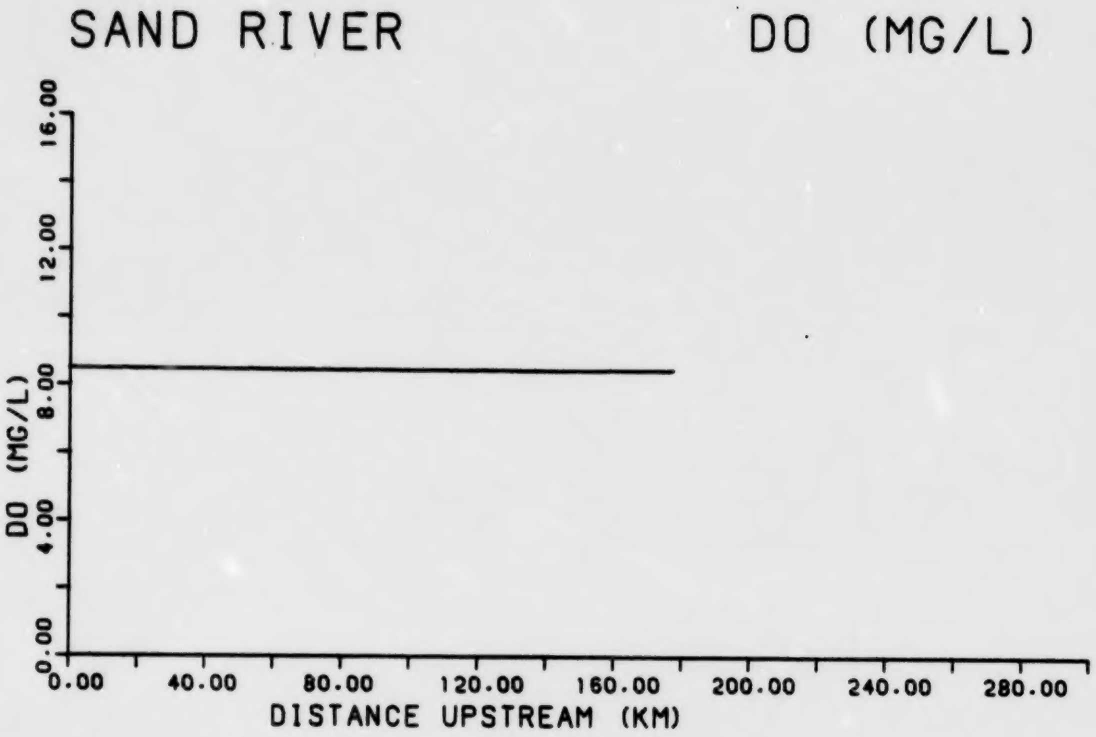

BEAVER RIVER
COLI (MG/L)
COLI (MG/L)
0.00
40.00
80.00
120.00
160.00
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

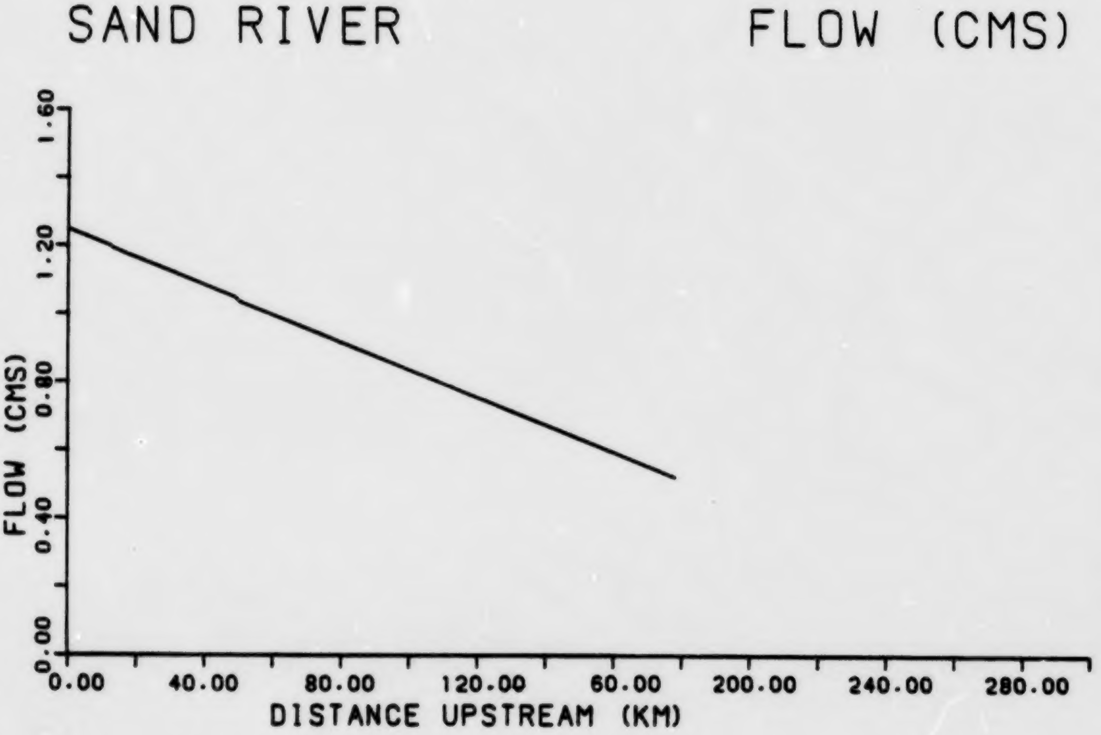

BEAVER RIVER
Na MG/L
Na MG/L
0.00
40.00
80.00
120.00
160.00
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

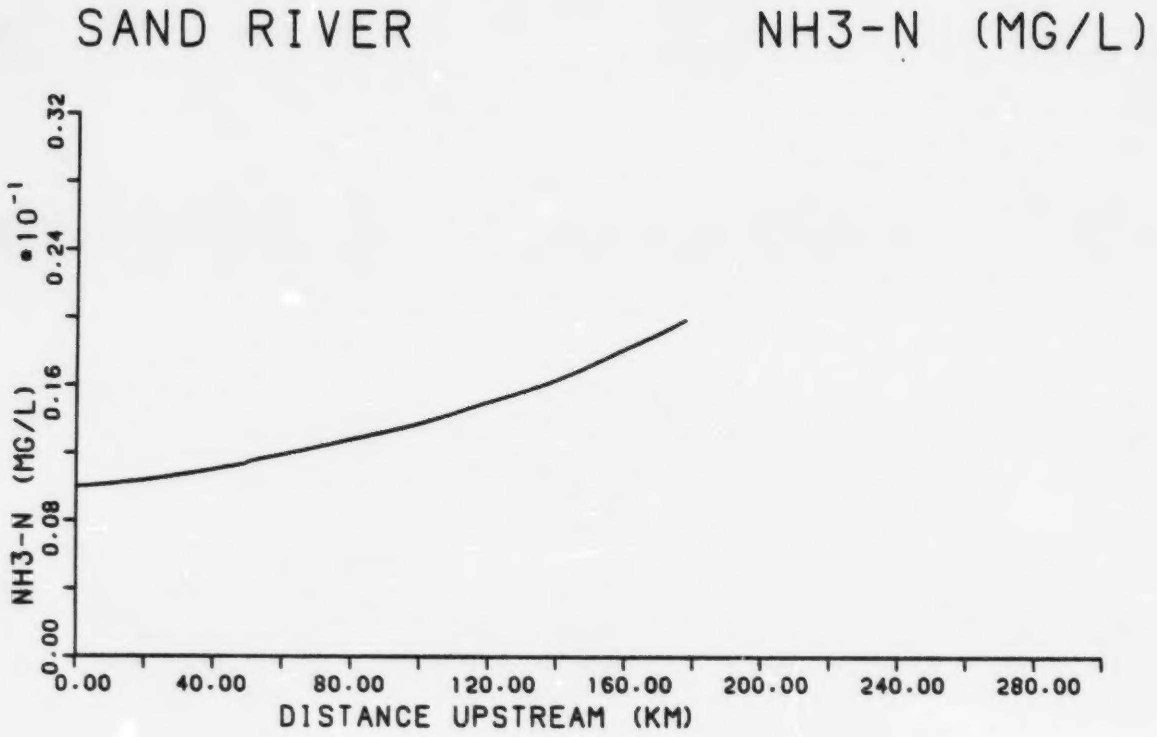

BEAVER RIVER
CL MG/L
CL MG/L
0.00
16.00
32.00
48.00
64.00
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

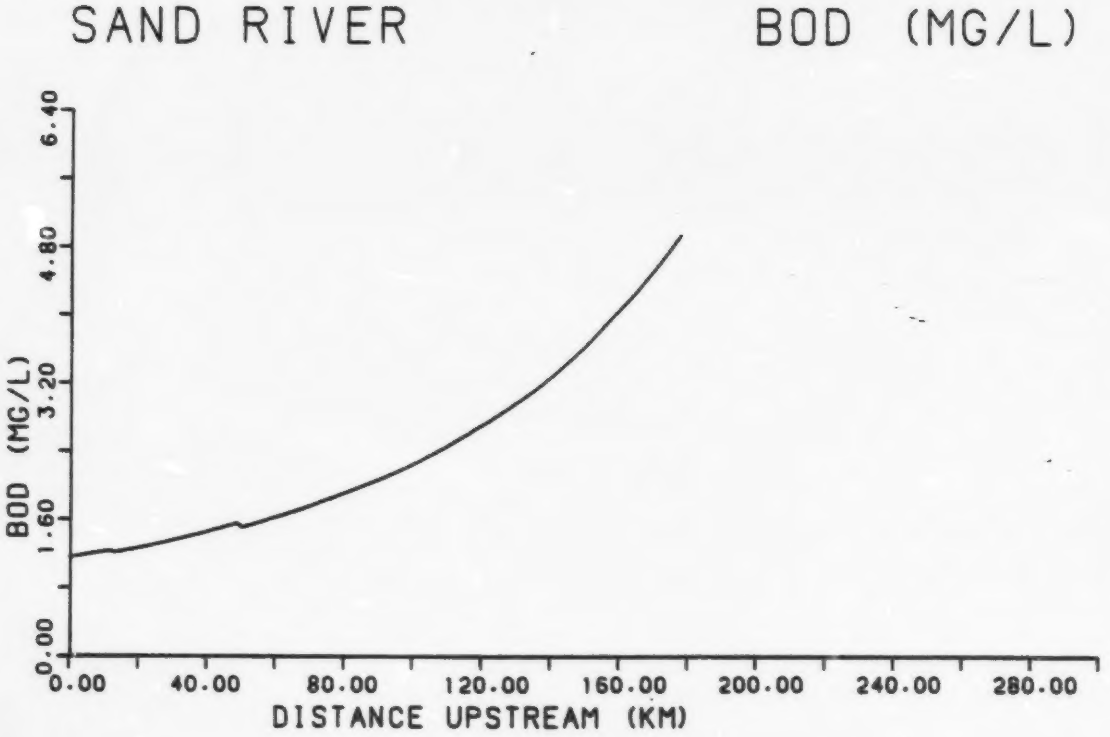

BEAVER RIVER
SO4 MG/L
SO4 MG/L
0.00
40.00
80.00
120.00
160.00
0.00
20.00
40.00
60.00
80.00
100.00
120.00
140.00
DISTANCE UPSTREAM (KM)

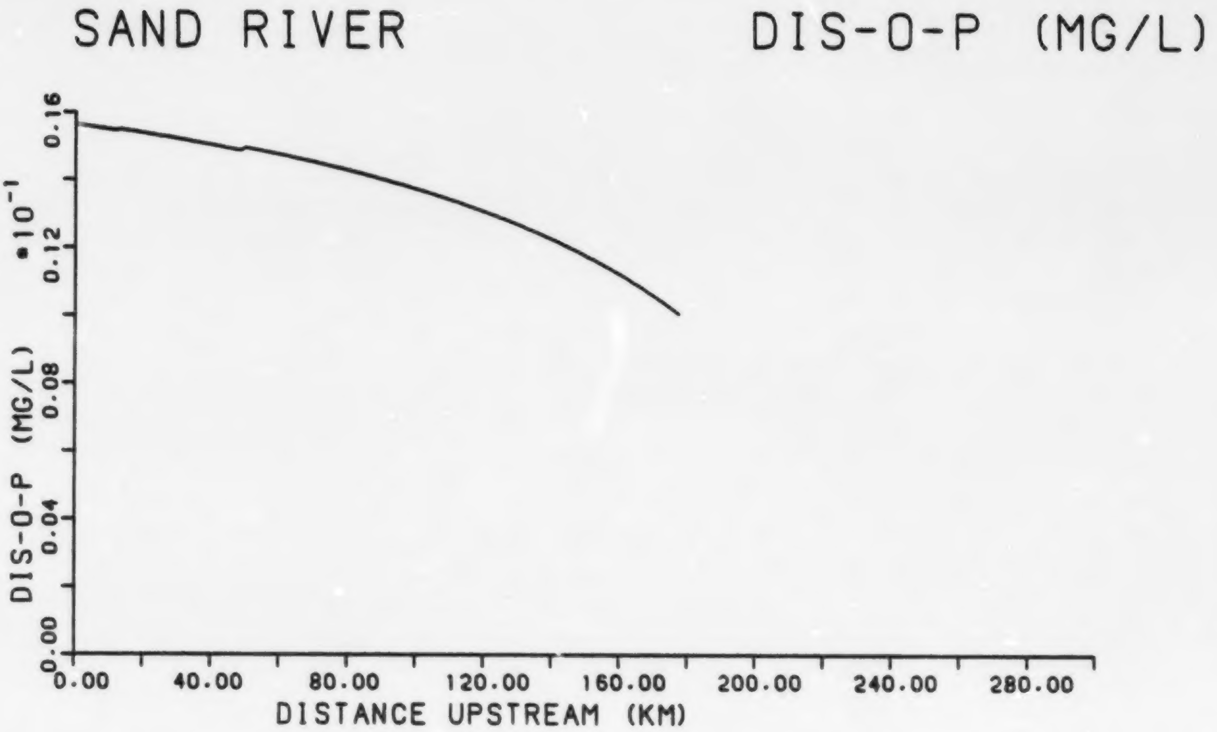
SAND RIVER

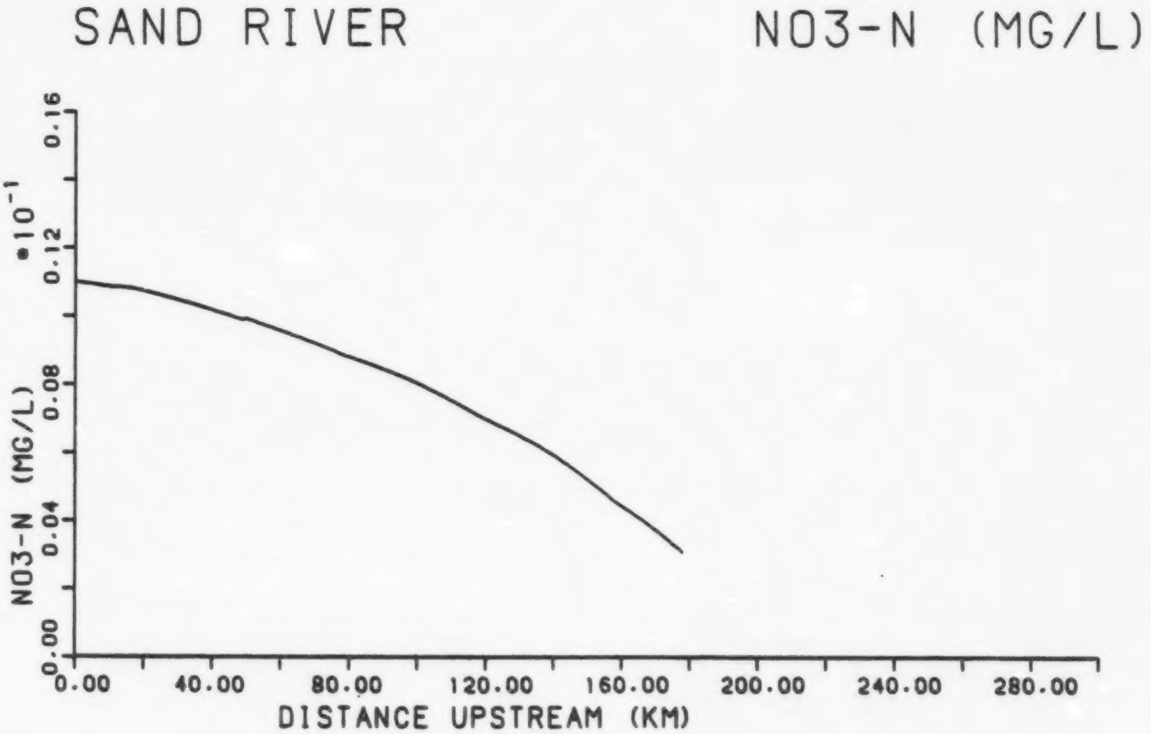

SAND RIVER
DO (MG/L)
DO (MG/L)
0.00
4.00
8.00
12.00
16.00
0.00
40.00
80.00
120.00
160.00
200.00
240.00
280.00
DISTANCE UPSTREAM (KM)


SAND RIVER
FLOW (CMS)
FLOW (CMS)
0.00
0.40
0.80
1.20
1.60
0.00
40.00
80.00
120.00
60.00
200.00
240.00
280.00
DISTANCE UPSTREAM (KM)

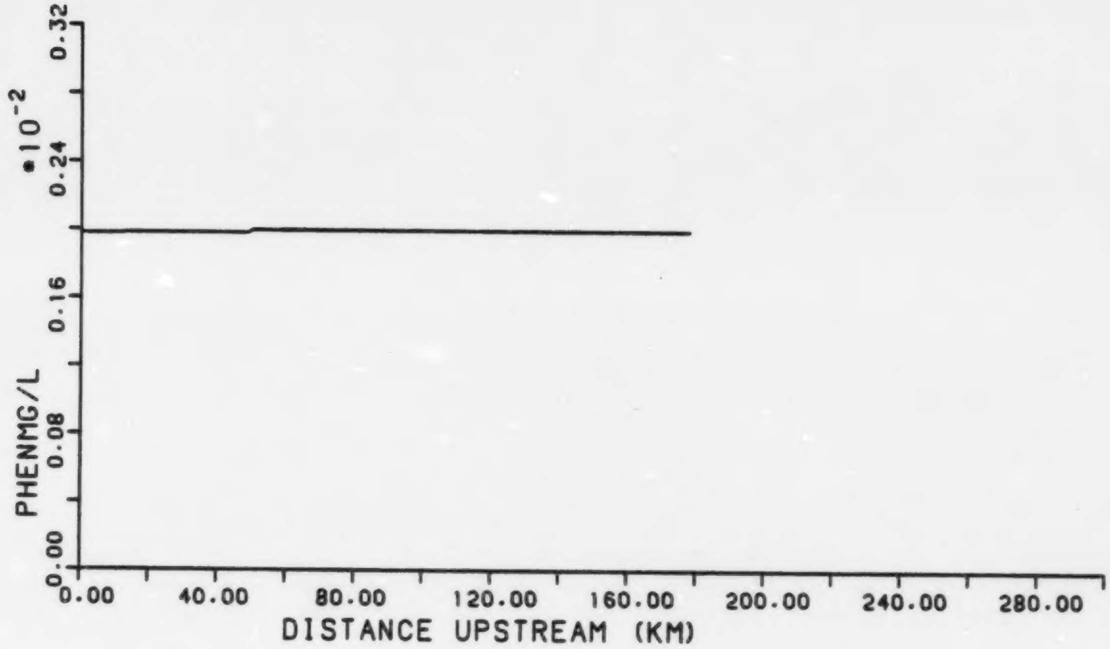

SAND RIVER
NH3-N (MG/L)
NH3-N (MG/L)
*10⁻¹
0.00
0.08
0.16
0.24
0.32
0.00
40.00
80.00
120.00
160.00
200.00
240.00
280.00
DISTANCE UPSTREAM (KM)


SAND RIVER
BOD (MG/L)
BOD (MG/L)
0.00
1.60
3.20
4.80
6.40
0.00
40.00
80.00
120.00
160.00
200.00
240.00
280.00
DISTANCE UPSTREAM (KM)

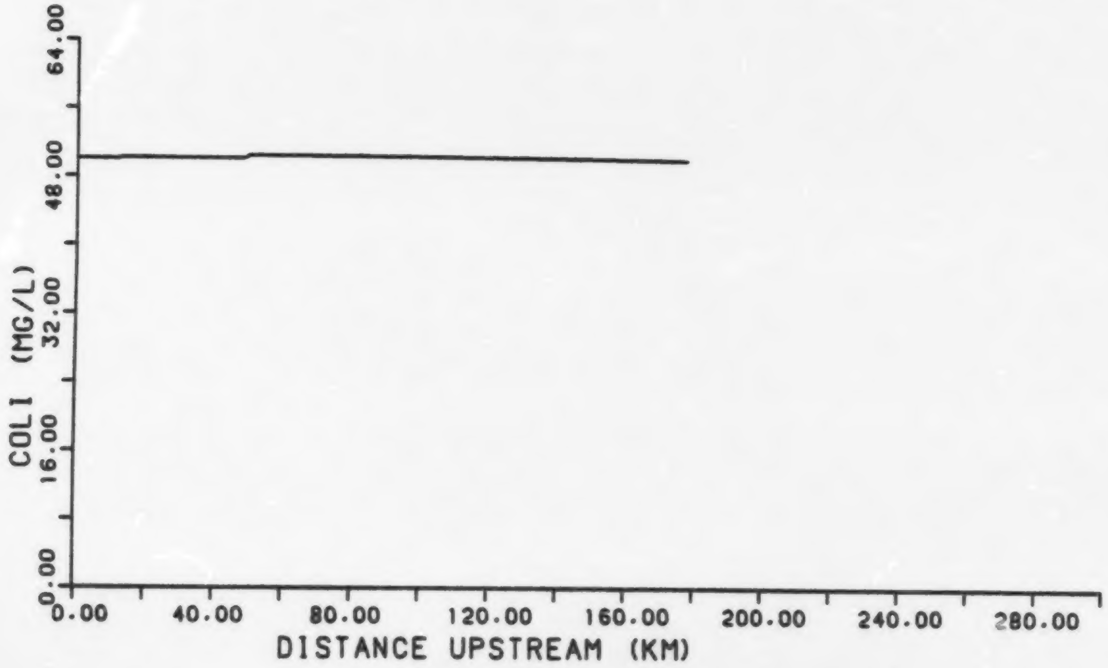

SAND RIVER
DIS-O-P (MG/L)
DIS-O-P (MG/L) *10^-1
0.00
0.04
0.08
0.12
0.16
0.00
40.00
80.00
120.00
160.00
200.00
240.00
280.00
DISTANCE UPSTREAM (KM)

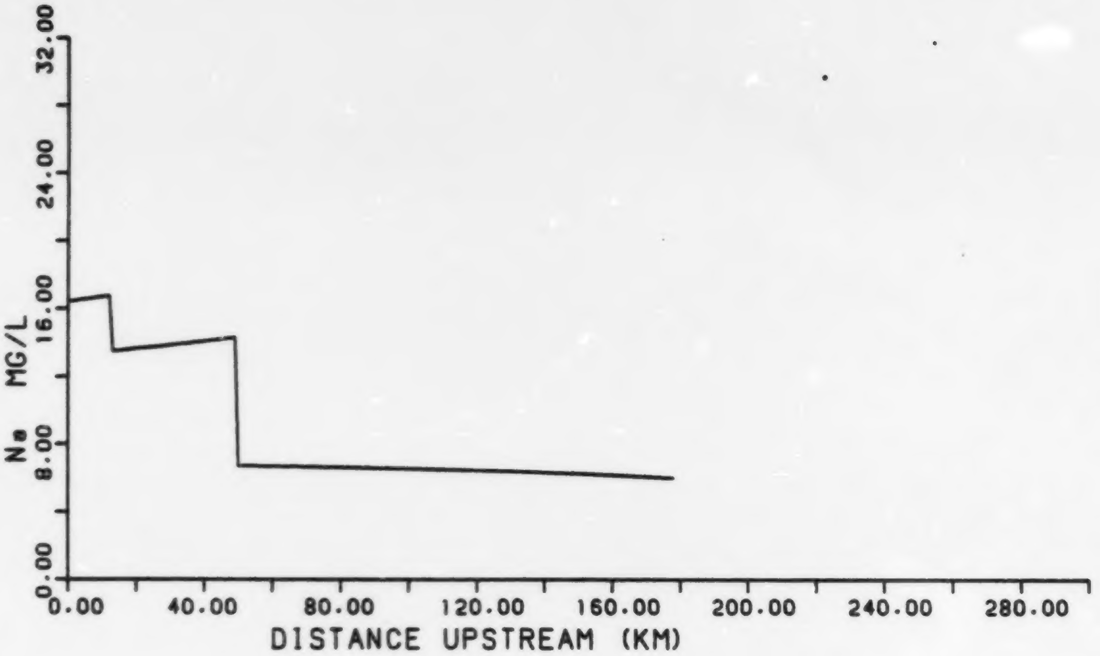

SAND RIVER
NO3-N (MG/L)
NO3-N (MG/L) *10^-1
0.00
0.04
0.08
0.12
0.16
0.00
40.00
80.00
120.00
160.00
200.00
240.00
280.00
DISTANCE UPSTREAM (KM)

SAND RIVER PHENMG/L

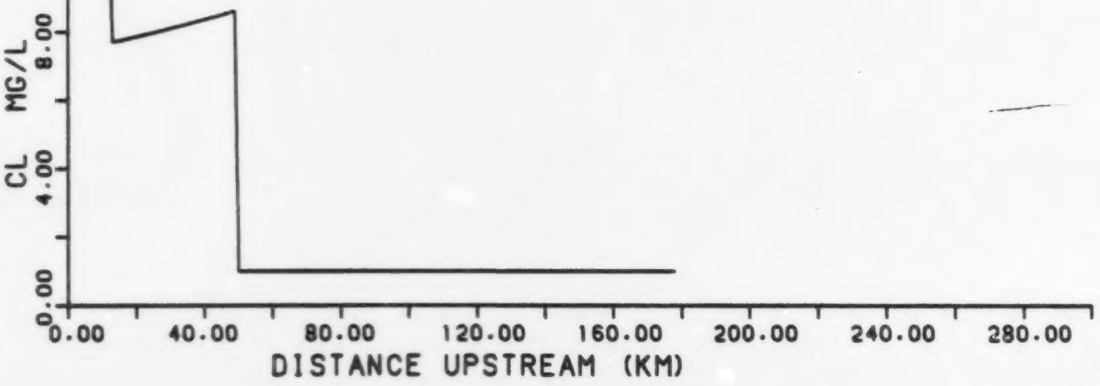


SAND RIVER COLI (MG/L)

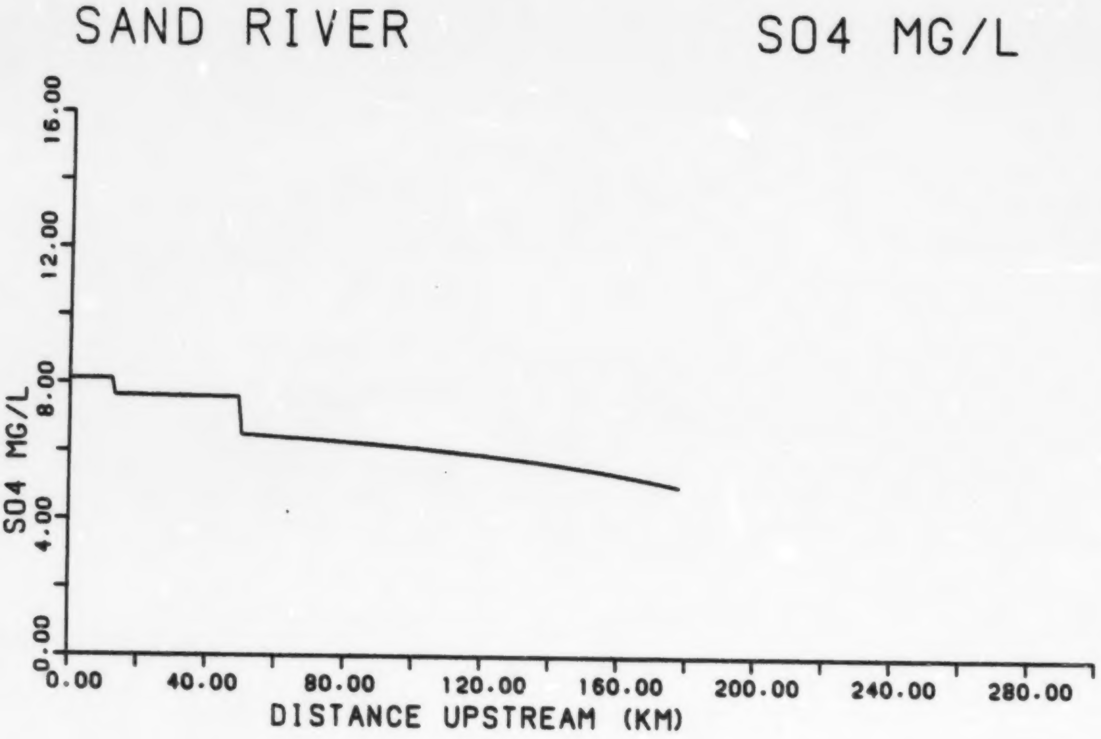




SAND RIVER Na MG/L

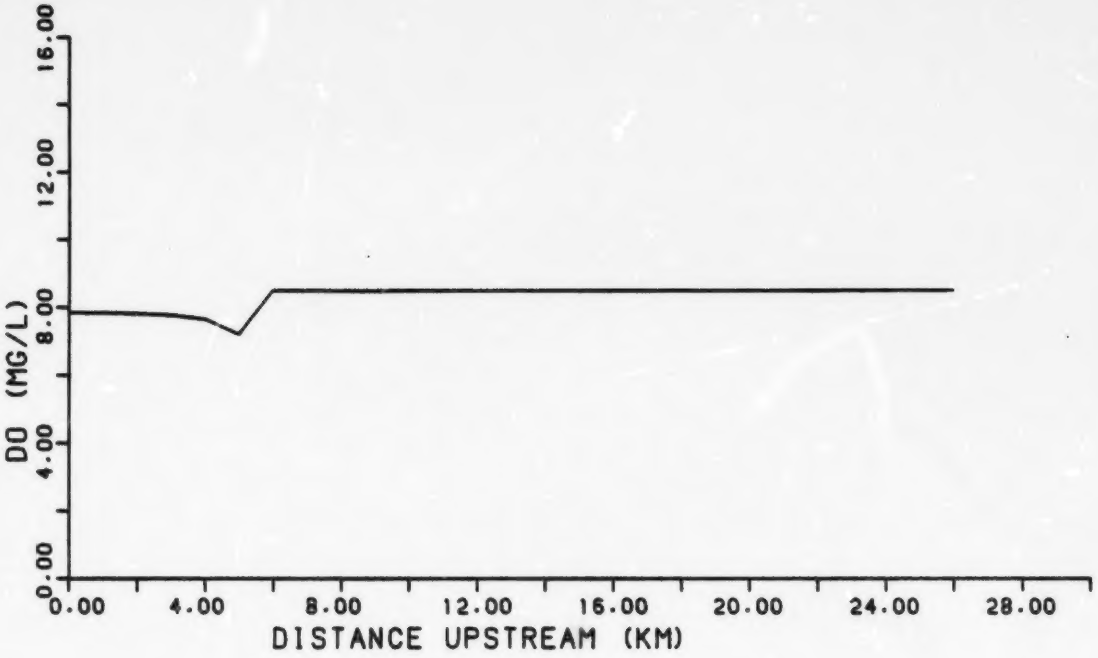


SAND RIVER CL MG/L

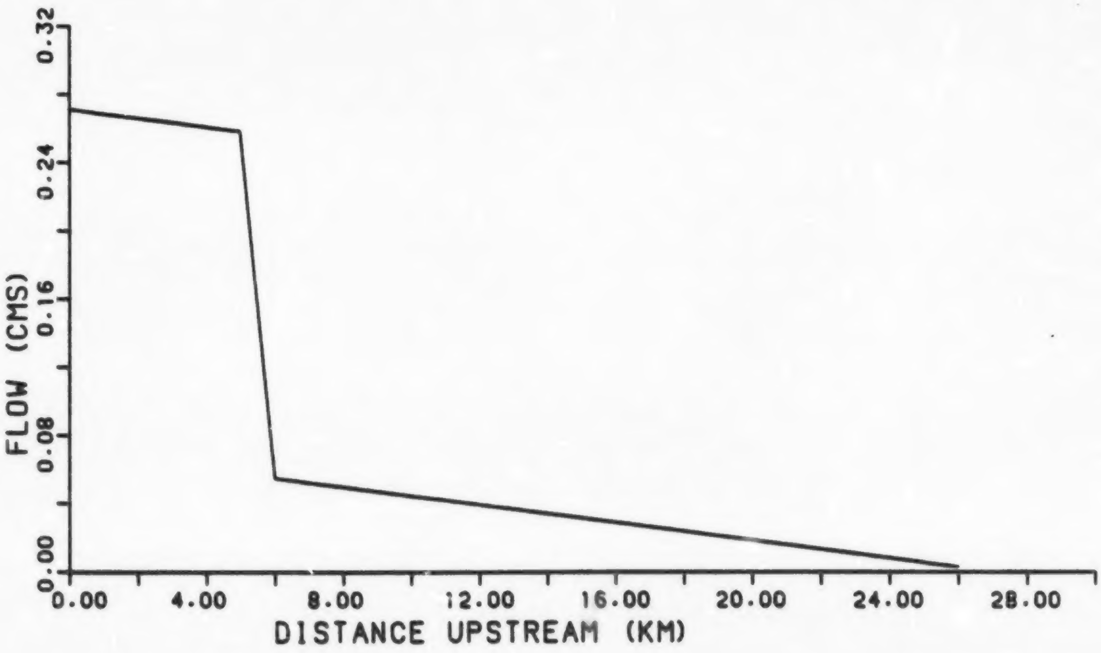

SAND RIVER
SO4 MG/L
SO4 MG/L
0.00
4.00
8.00
12.00
16.00
0.00
40.00
80.00
120.00
160.00
200.00
240.00
280.00
DISTANCE UPSTREAM (KM)

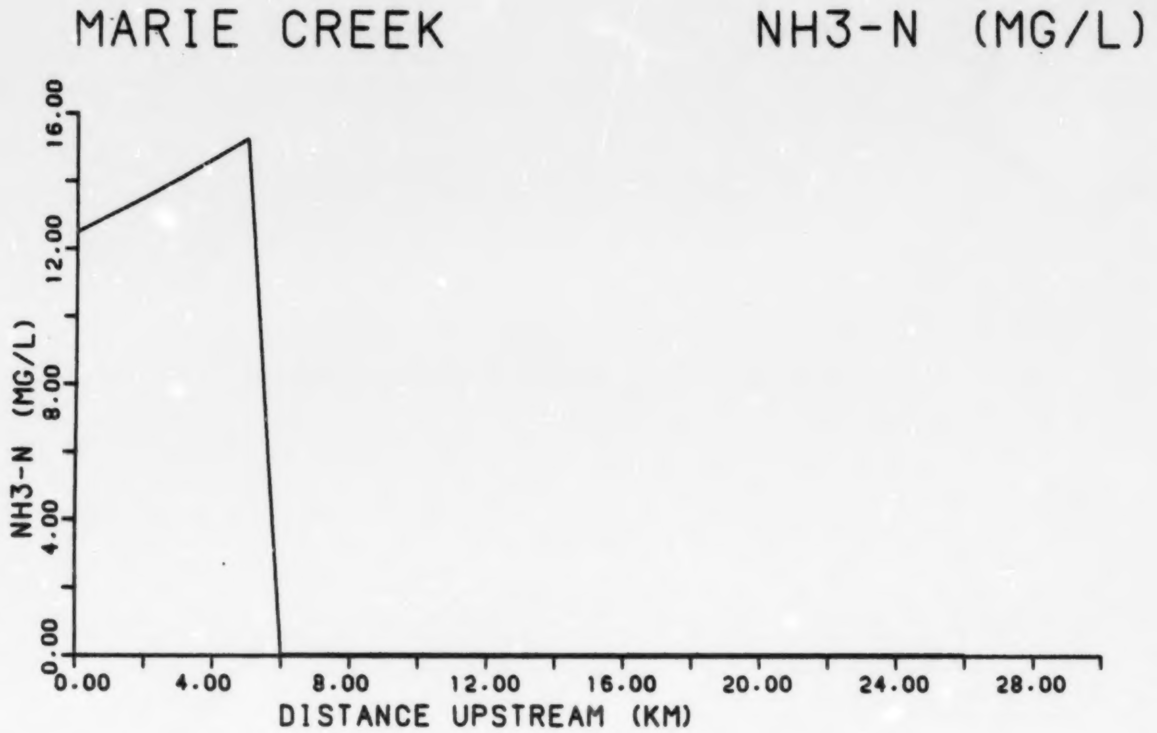
MARIE CREEK

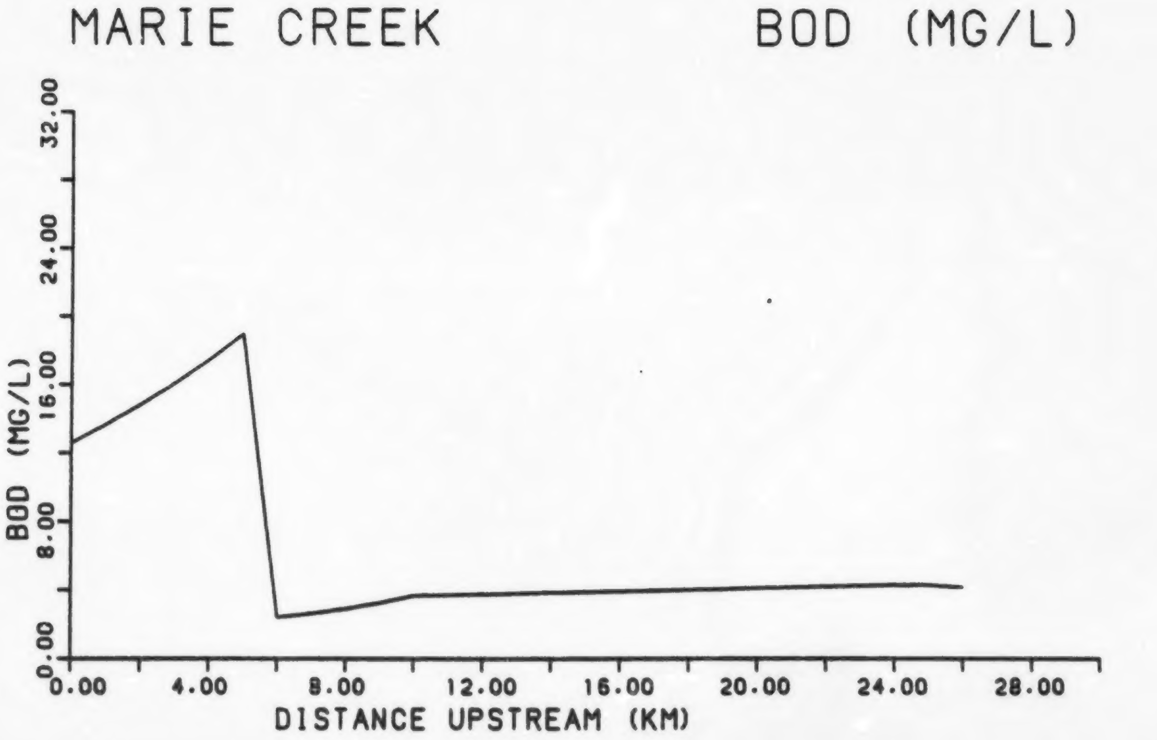
MARIE CREEK DO (MG/L)



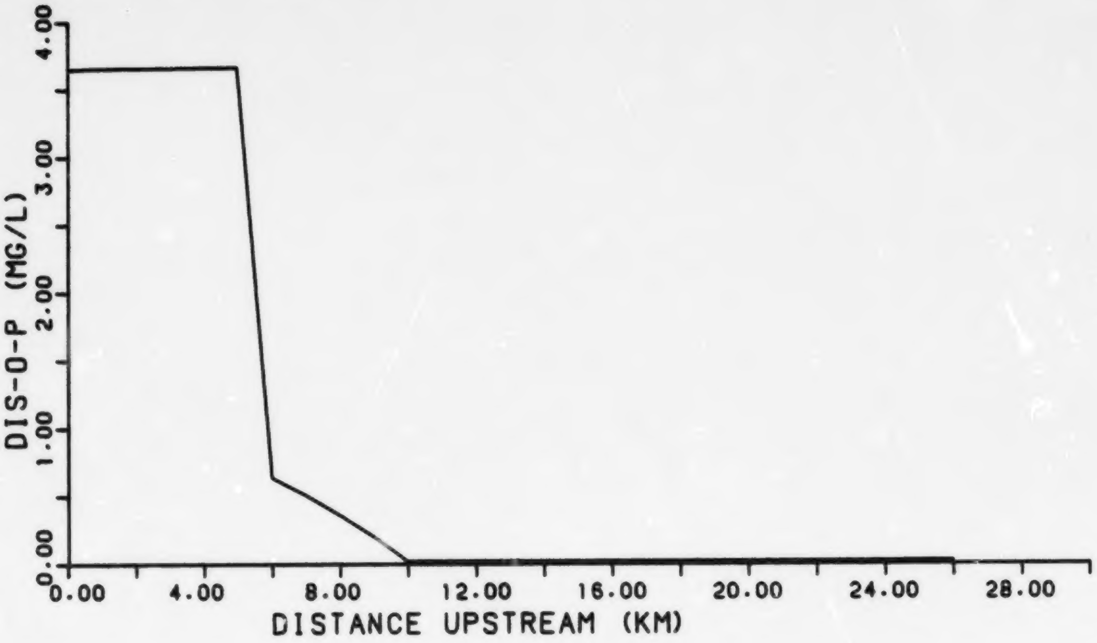
MARIE CREEK FLOW (CMS)



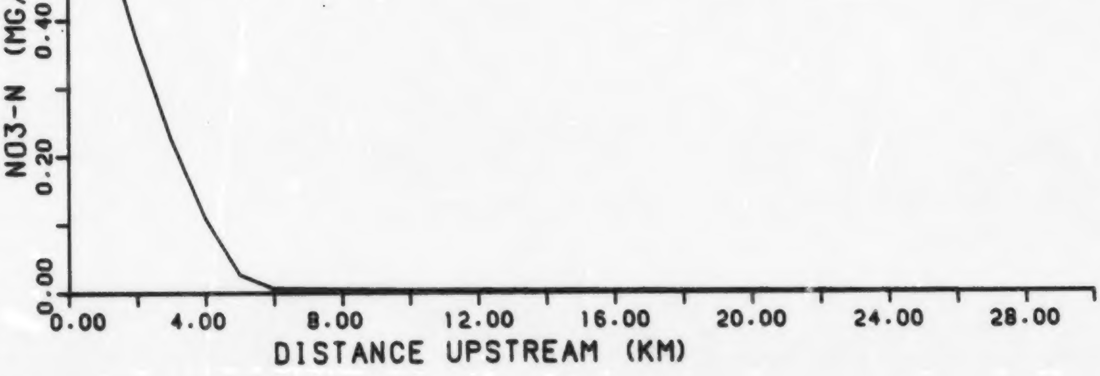

MARIE CREEK
NH3-N (MG/L)
NH3-N (MG/L)
0.00
4.00
8.00
12.00
16.00
0.00
4.00
8.00
12.00
16.00
20.00
24.00
28.00
DISTANCE UPSTREAM (KM)

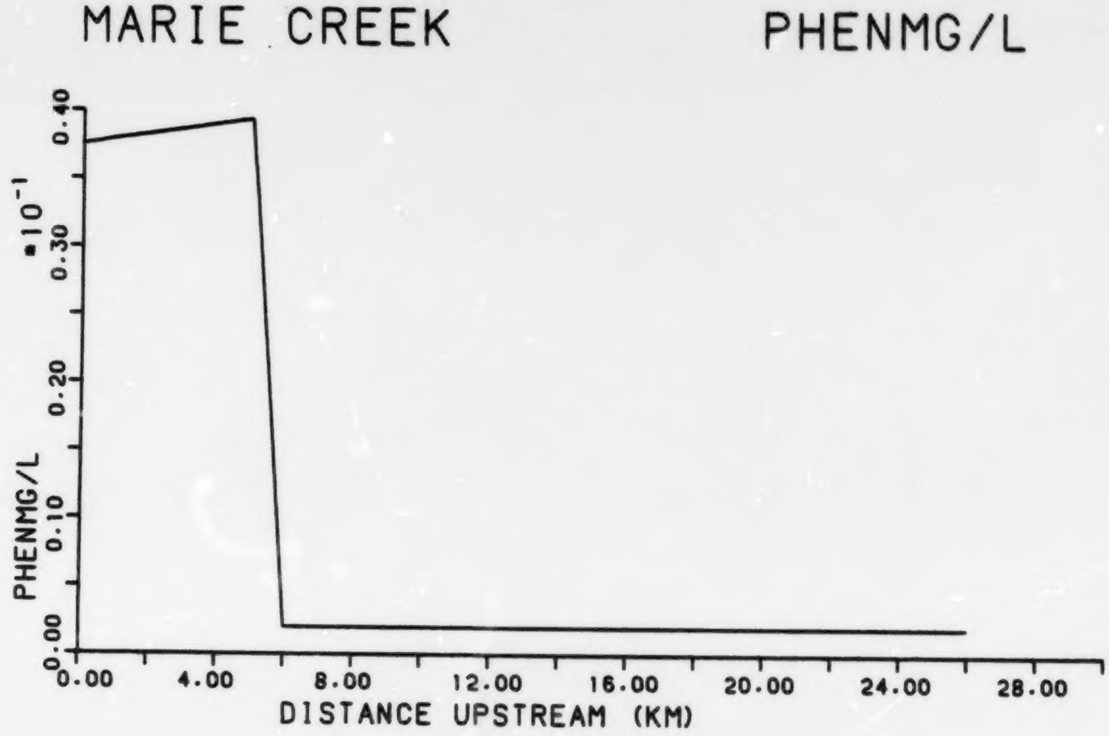

MARIE CREEK
BOD (MG/L)
BOD (MG/L)
0.00
8.00
16.00
24.00
32.00
0.00
4.00
8.00
12.00
16.00
20.00
24.00
28.00
DISTANCE UPSTREAM (KM)

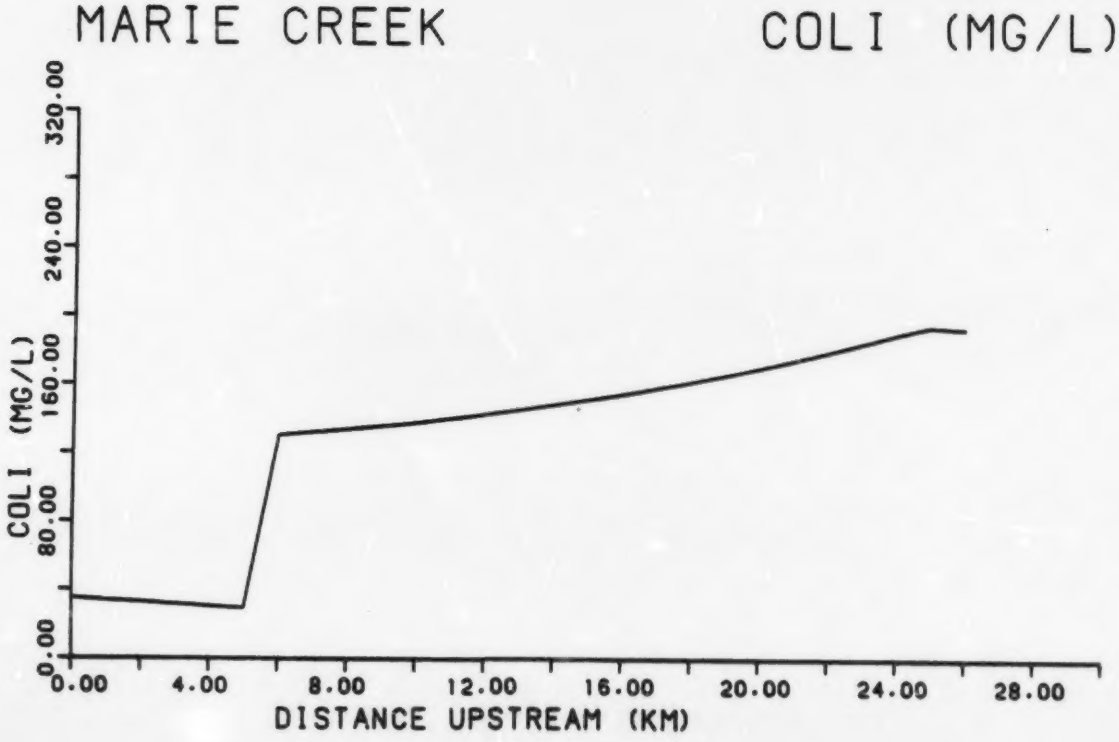
MARIE CREEK DIS-O-P (MG/L)



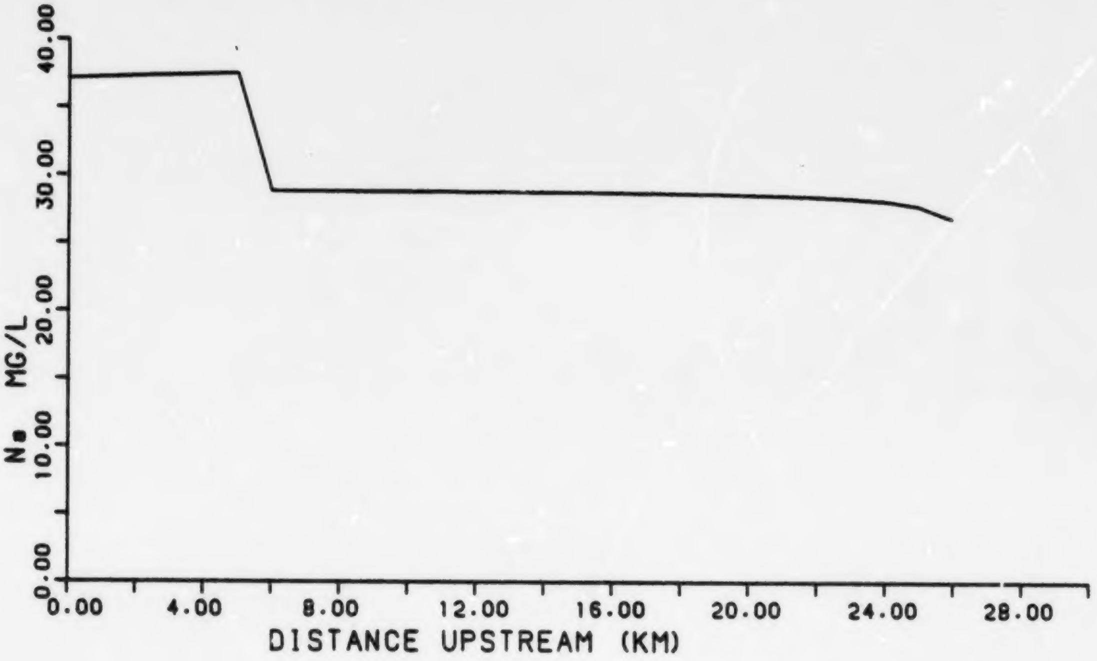
MARIE CREEK NO3-N (MG/L)



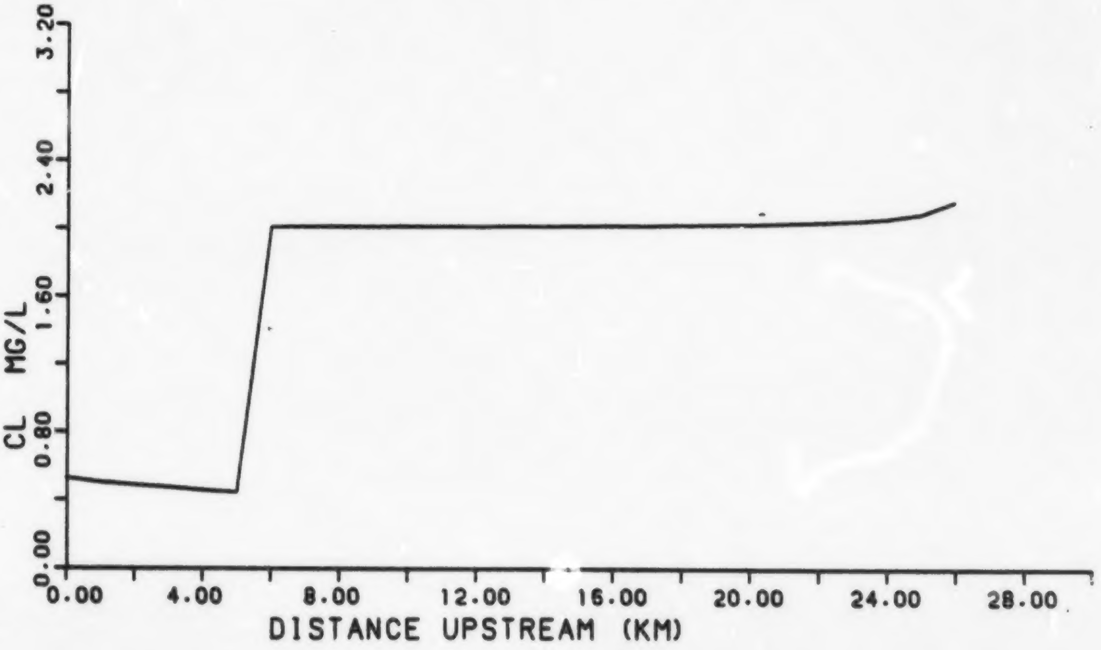

MARIE CREEK
PHENMG/L
*10⁻¹
PHENMG/L
0.00
0.10
0.20
0.30
0.40
0.00
4.00
8.00
12.00
16.00
20.00
24.00
28.00
DISTANCE UPSTREAM (KM)


MARIE CREEK
COLI (MG/L)
COLI (MG/L)
0.00
80.00
160.00
240.00
320.00
0.00
4.00
8.00
12.00
16.00
20.00
24.00
28.00
DISTANCE UPSTREAM (KM)

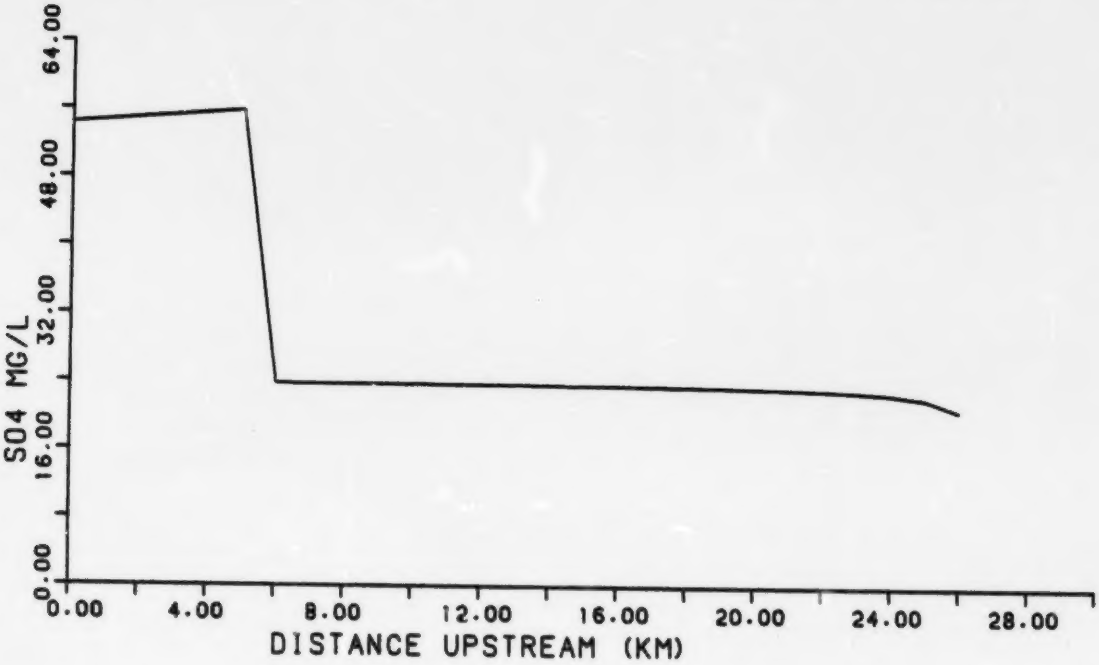
MARIE CREEK Na MG/L



MARIE CREEK CL MG/L

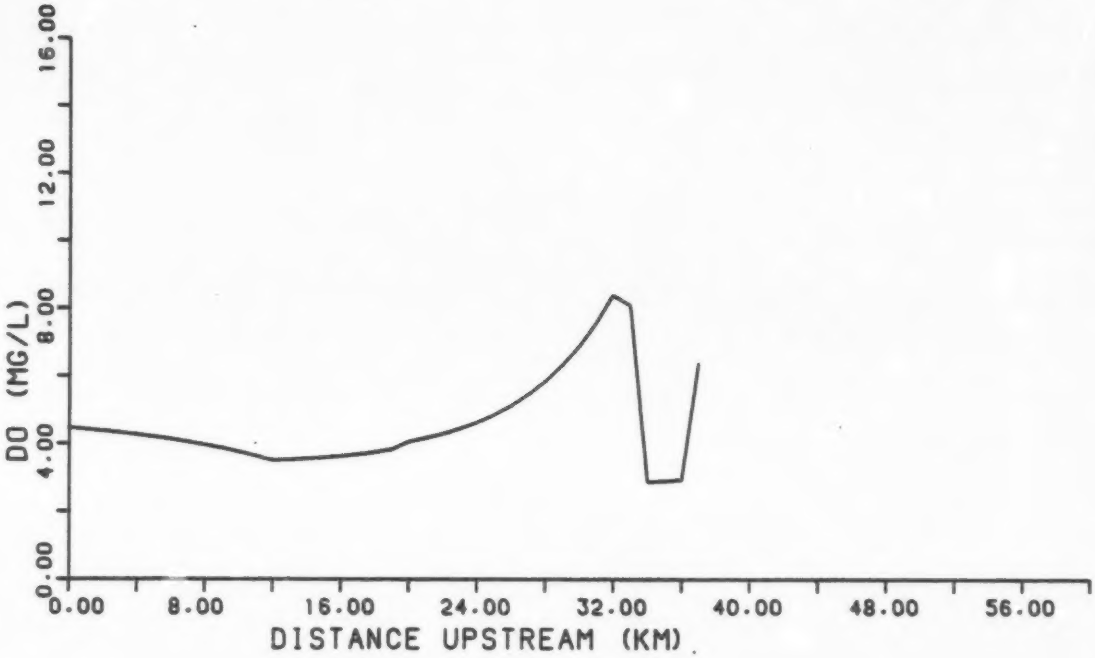


MARIE CREEK SO4 MG/L

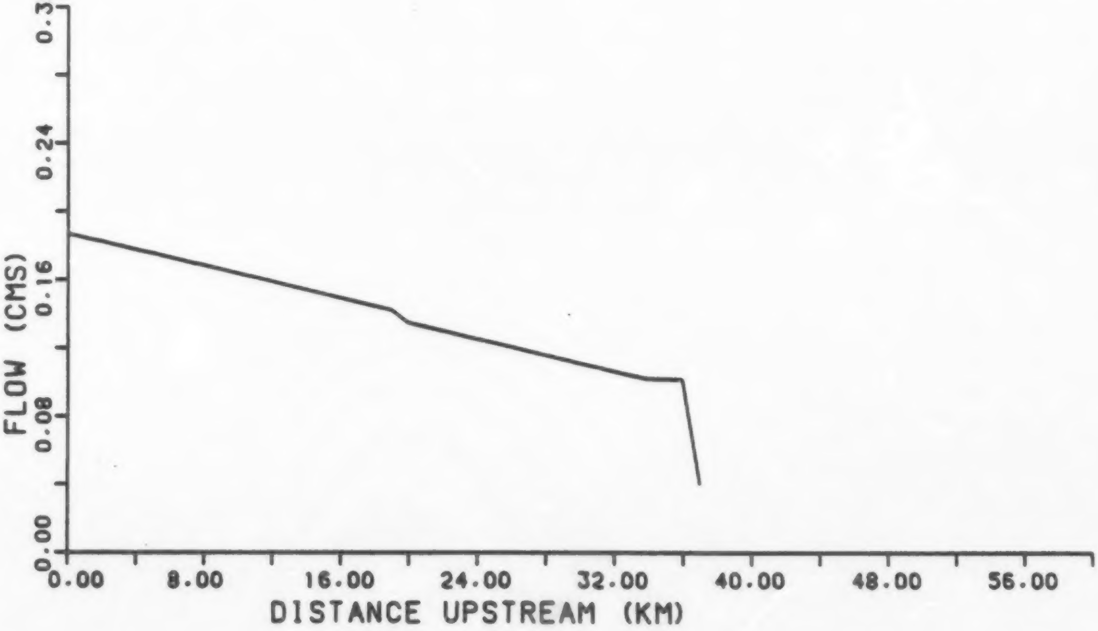


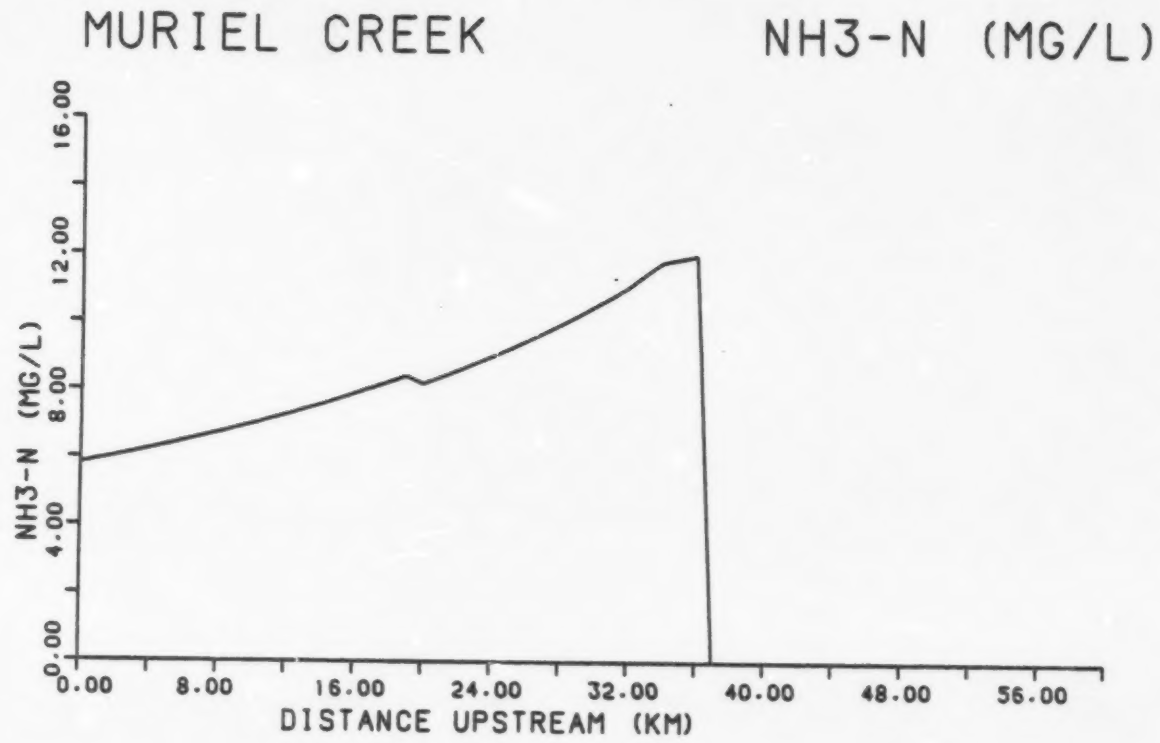
MURIEL CREEK

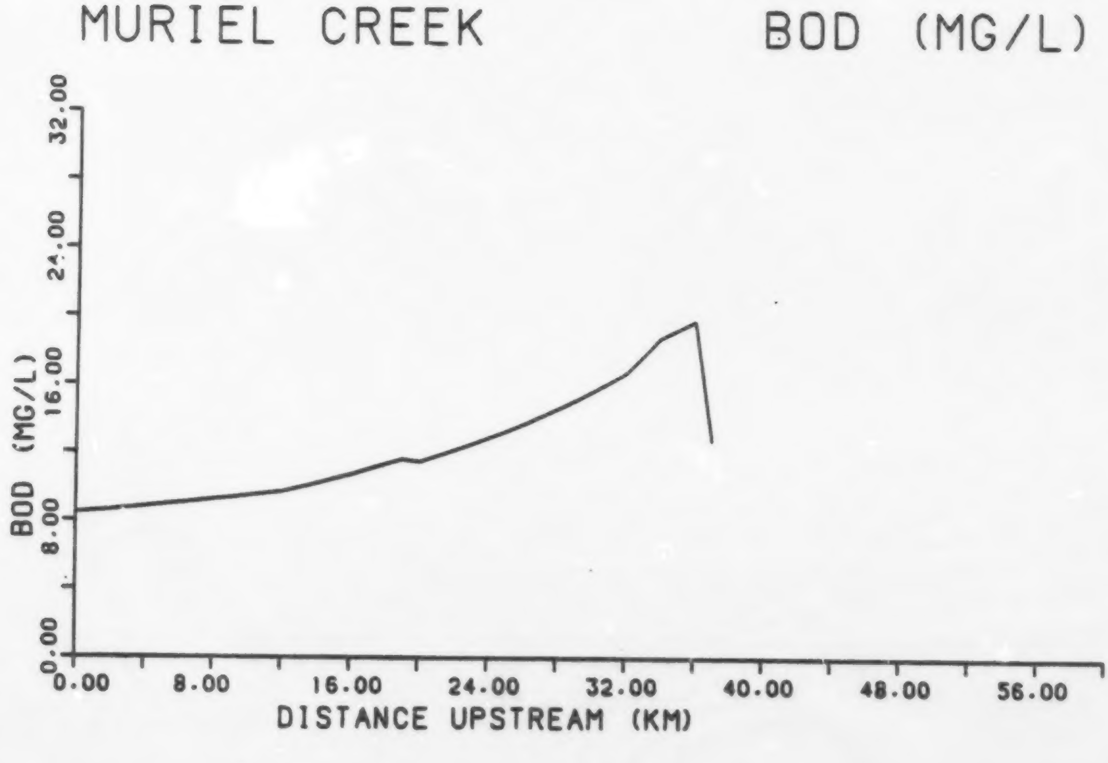
MURIEL CREEK DO (MG/L)



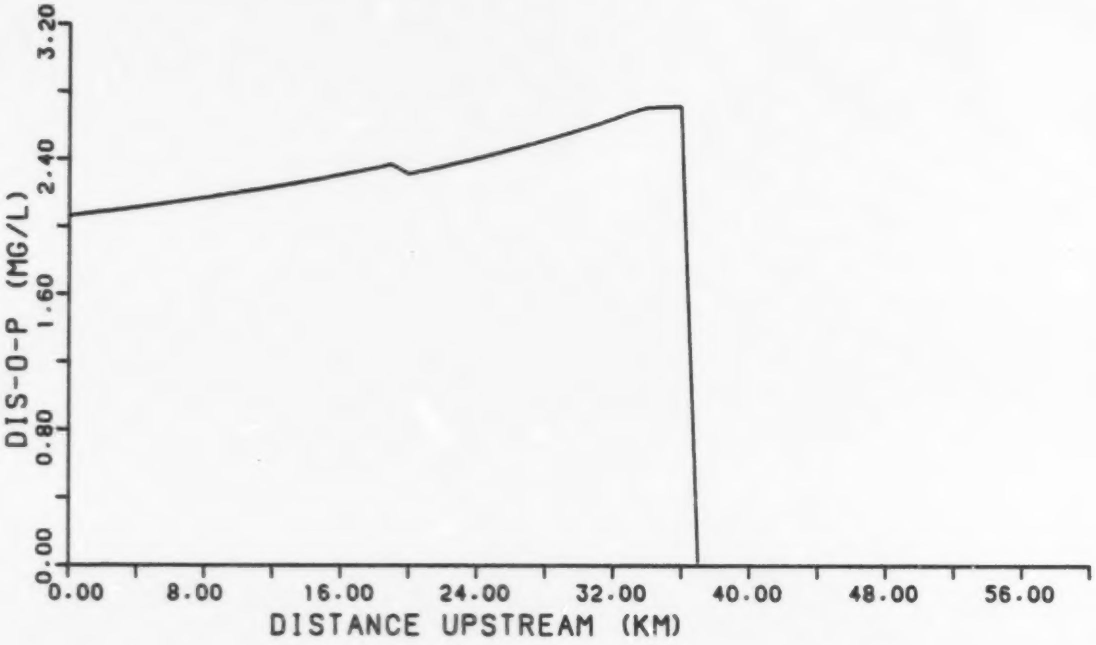
MURIEL CREEK FLOW (CMS)



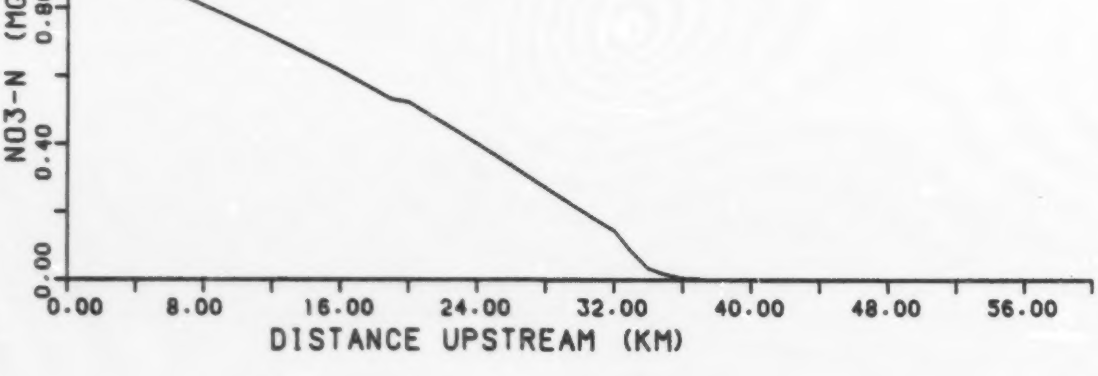

MURIEL CREEK
NH3-N (MG/L)
NH3-N (MG/L)
0.00
4.00
8.00
12.00
16.00
0.00
8.00
16.00
24.00
32.00
40.00
48.00
56.00
DISTANCE UPSTREAM (KM)


MURIEL CREEK
BOD (MG/L)
BOD (MG/L)
0.00
8.00
16.00
24.00
32.00
0.00
8.00
16.00
24.00
32.00
40.00
48.00
56.00
DISTANCE UPSTREAM (KM)

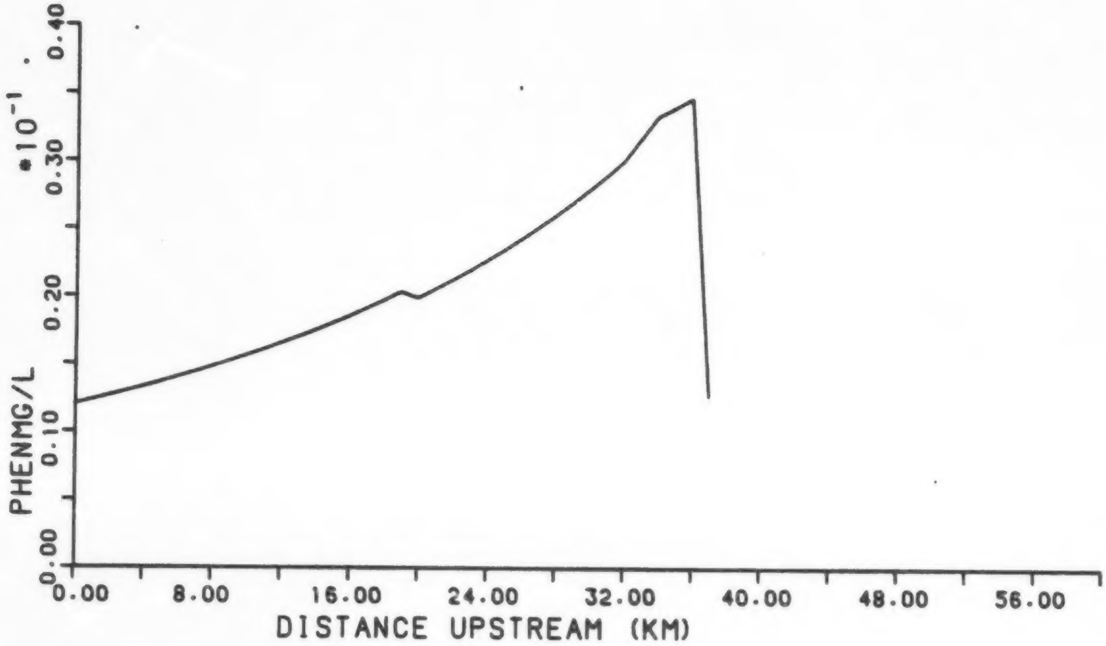
MURIEL CREEK DIS-O-P (MG/L)



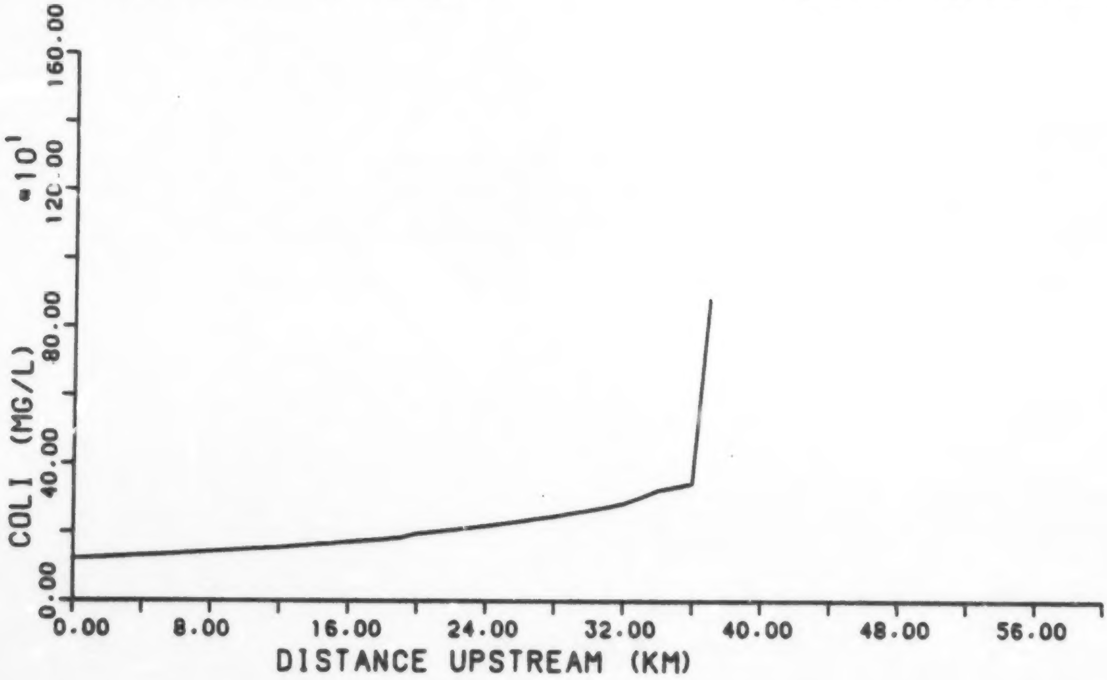
MURIEL CREEK NO3-N (MG/L)

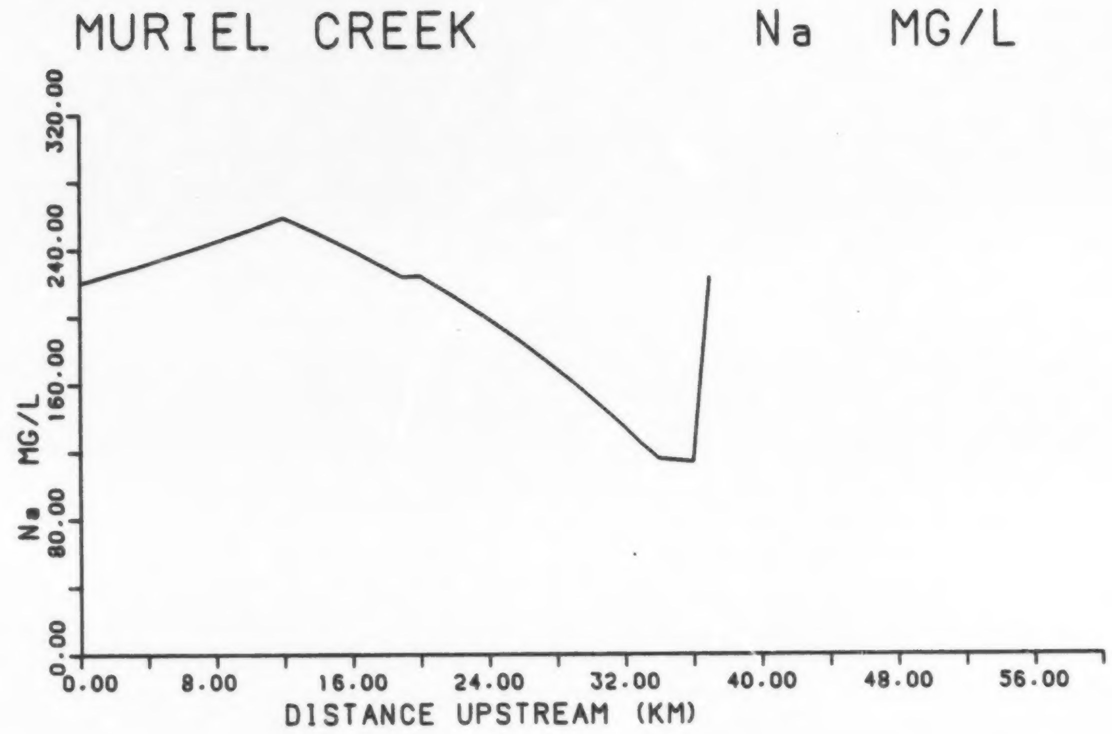


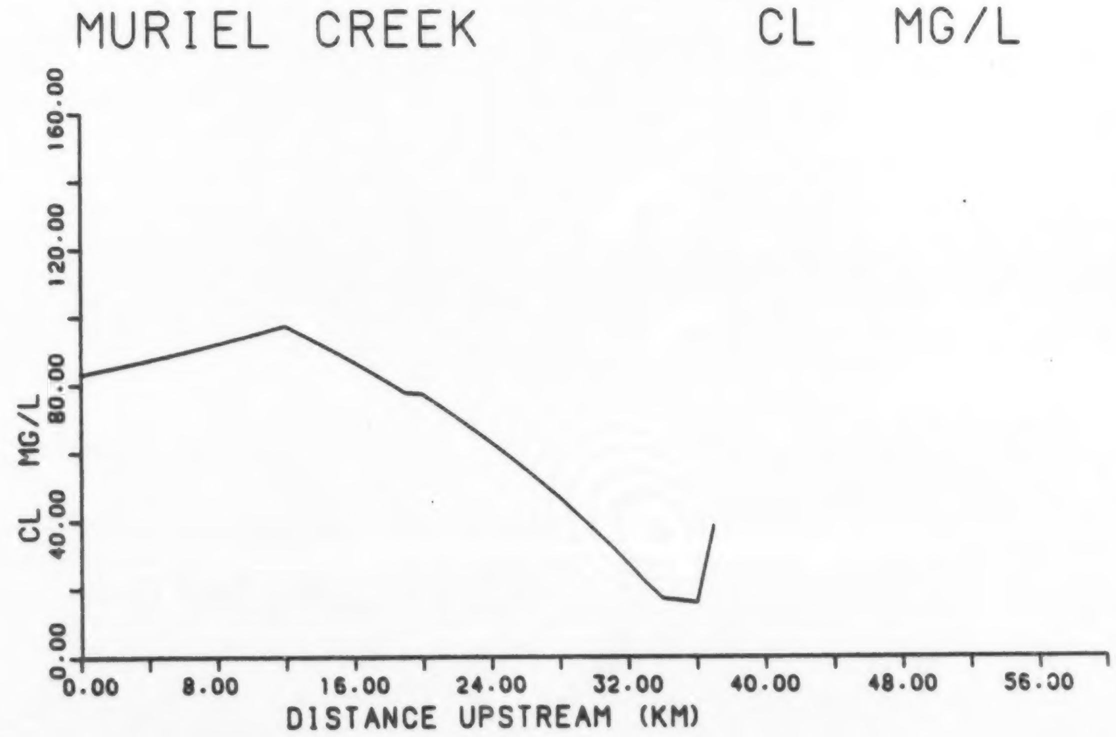
MURIEL CREEK PHENMG/L



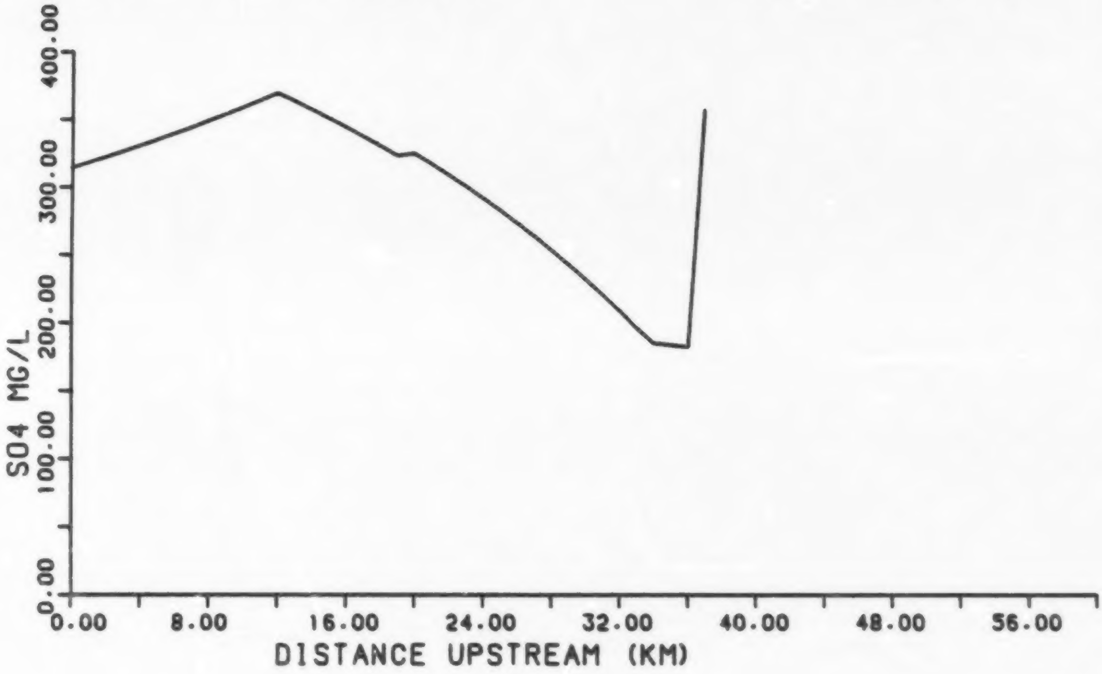
MURIEL CREEK COLI (MG/L)




MURIEL CREEK
Na MG/L
320.00
240.00
160.00
80.00
0.00
Na MG/L
0.00
8.00
16.00
24.00
32.00
40.00
48.00
56.00
DISTANCE UPSTREAM (KM)


MURIEL CREEK
CL MG/L
160.00
120.00
80.00
40.00
0.00
CL MG/L
0.00
8.00
16.00
24.00
32.00
40.00
48.00
56.00
DISTANCE UPSTREAM (KM)

MURIEL CREEK SO4 MG/L



POOR

QUALITY

ORIGINAL